

**Methods for Monitoring the Abundance of the Northern  
Australian Mud Crab *Scylla serrata***

By

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## ABSTRACT

The final outcomes of the 1996 Northern Territory mud crab (*Scylla serrata*) fishery assessment recommended the development of alternate methods for monitoring the abundance and habitat of this species; citing difficulties with analyses of simple CPUE data. In particular the assessment team recommended the testing of a model where:

$$\text{Abundance} = \text{area of critical habitat} \times \text{density of animals per unit of habitat}$$

Determination of the suitability of this model required the estimation of two parameters: the area of each critical habitat type and the abundance of crabs per unit of critical habitat type.

Consultation with fishery stakeholders, review of historical fishery information and a survey of the literature provided information on the key habitat associations of the mud crab. The two habitats of primary interest, mangroves and mud flats, are generally remote and difficult to access in northern Australia. An existing regional wetland dataset collected using remote-sensing techniques (Landsat ETM+ imagery) was identified and additional analyses were completed providing estimates of area of mangrove, mangrove lined waterways, and mud flat foreshore for each fishing grid.

Methods to estimate mud crab abundance within habitat type were reviewed for fitness against species behaviour and environmental conditions. Due to the dynamic nature of this fishery my initial work focused on gaining an understanding of the potential uncertainties in meeting the analysis assumptions. Preliminary studies were undertaken to test various assumptions and uncertainties, such as environmental and biological effects on catchability. An examination of crab movement patterns during short term study periods demonstrated little evidence of large scale movement and a general acceptance of the assumption of a closed population.

Mark recapture techniques previously utilised in terrestrial animal abundance estimation were applied for the first time in a marine environment. A new study approach incorporating depletion and mark recapture methods was utilised for mangrove-lined streams and a novel trapping web design for the foreshore mudflat areas. The biomass estimates obtained from these studies were then assessed against four commonly used fishery assessment metrics, commercial catch, commercial effort, commercial CPUE, fishery independent CPUE. Commercial CPUE performed poorly and fishery independent CPUE data partially supported observed trends in the catch. These results provide support that CPUE data performs poorly in assessment of this fishery. Biomass estimates obtained from the mark recapture studies demonstrated very similar temporal trends at both study sites providing good support for the fishery independent methods tested in this project.

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# 1. INTRODUCTION

The northern Australian commercial mud crab (*Scylla serrata*) fishery was valued at approximately A\$20 million in 2000 (Calogeras 2000) with the majority of commercial fishing activity for mud crabs occurring within Northern Territory and Queensland waters. Mud crabs are also an important focus of the recreational sectors particularly in Queensland where the recreational catch is thought to be at least equal to the Queensland commercial catch. The mud crab is also a significant food source for northern Australian coastal Aboriginal communities.

Northern Australian mud crab fisheries are highly regulated. In the Northern Territory regulations for the commercial industry include the use of both input and output controls. A total of 49 mud crab licences are issued annually and each licence holder is entitled to use a maximum of 60 traps. Output controls are also applied with Northern Territory size limits set at a minimum of 130 mm carapace width (CW) for male and 140 mm CW for female mud crabs. Northern Territory recreational fishing controls include a possession limit of 10 crabs per angler or 30 crabs per boat, a total of 10 crab traps may be used and size limits are the same as the commercial fishery.

A 'Northern Australia Mud Crab Research Priority Workshop' held in Darwin during May 1999 was attended by researchers, managers and key industry representatives from the Northern Territory, Queensland, and Western Australia. The major outcome from the workshop was the development of a five-year research plan for northern Australian mud crab fisheries, based on both industry and management issues (Calogeras 2000). In order to maximise the benefits of available resources it was decided to adopt a phased 5-year approach. The three phases were:

1. Examination of the relative productivity of mud crab habitat types.
2. Comparison of stock abundance indicators.
3. Investigation the spatial differences in population reproductive characteristics.

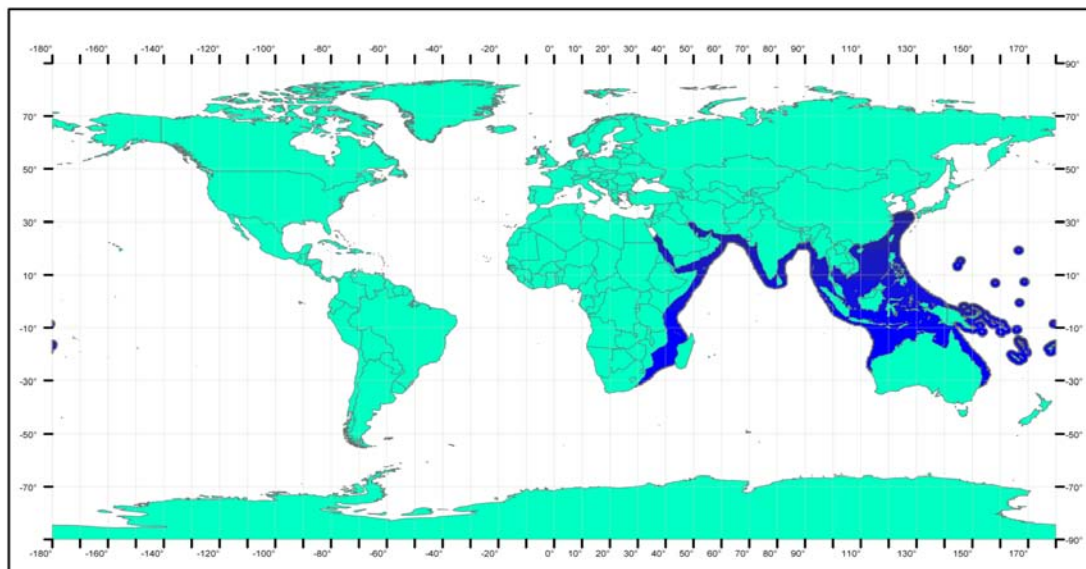
The work reported in this thesis addresses the priorities identified for phase one of the research plan. The major objective of this research was:

To evaluate new fishery independent methods for obtaining mud crab biomass. Specifically, I tested a combined mark-recapture and depletion method for river systems and a trapping web method for mudflat foreshores so as to cover both major fishable mud crab habitats.

## 2. LITERATURE REVIEW

### 2.1 Mud Crabs

The portunid crab, *Scylla serrata*, occurs throughout much of the inshore regions of the Indo-West Pacific Ocean (Hill *et al.* 1982a) (Figure 1). Within this broad geographical distribution, *S. serrata* is commonly found within mangrove swamps, estuaries and sheltered parts of the coastal shoreline (Hill 1975). It strongly prefers muddy substrates, which contributes to its common name as the mud or mangrove crab (Brown 1993).



**Figure 1.** Distribution *Scylla serrata* Indo-West Pacific: From east and South Africa to southeast and east Asia, northern and subtropical Australia, Marianas, Fiji and the Samoa Islands.

Within Australian waters, *S. serrata* occurs from Exmouth, Western Australia (latitude 22°S) across the entire Northern Territory and Queensland coastlines and extends down the east coast to Port Jackson (New South Wales) (latitude 34°S) (Heasman 1980). Recent reports from Western Australia Fisheries (Bellchamber 2001) indicate the species has been found in estuaries surrounding Perth, but it is unclear whether this has resulted from migration or translocation.



**Figure 2.** Adult male *Scylla serrata*.

*S. serrata* is a large, aggressive, omnivorous scavenger (Figure 2) with strong cannibalistic tendencies (Arriola 1940). This agonistic behaviour provides substantial incentive for individual animals to avoid each other (Robertson 1989) and disperse over the available habitat. Because of this, the mud crab utilises a variety of inshore habitats and environmental conditions during its life cycle, and these must be understood in order to manage the species in a sustainable manner (Hill *et al.* 1982b).

## 2.2 Taxonomy

The taxonomy of the genus *Scylla* has provoked much discussion. Until recently, numerous conflicting scientific observations debated the existence of between one and six additional species throughout the Indo-West Pacific. Much of this confusion can be attributed to the loss of the type specimen *Cancer serrata* (Forskål) collected from the Red Sea in 1775.

Estampador (1949) completed an important study in which he examined the external morphology of *Scylla sp.* sourced from the Philippines. He recognised the existence of three species and one subspecies of *Scylla*. Serene (1952) also noted four forms of *Scylla* in Vietnam, accepting the classification of Estampador. However, Stephenson and Campbell (1960) could only confirm the existence of one *Scylla* species from Queensland and New South Wales and suggested that observed morphological differences were derived from environmental conditions. Yet researchers from various Indo-Pacific regions continued to produce sound arguments for the existence of more than one species (Radhakrishnan and Samuel 1982; Fuseya and Watanabe 1996; Overton *et al.* 1997). While some authors followed the direction of Estampador, others formed their own interpretation of the taxonomy at hand. Problems in identification have been confounded by the considerable variation in colour and growth that occurs naturally across its broad geographical distribution (Perrine 1978) and the lack of a type specimen (Keenan *et al.* 1998). This uncertainty prompted Keenan *et al.* (1998) to conduct a taxonomic review of the genus.

The revision employed two independent genetic methods, allozyme electrophoresis (Keenan 1995; Keenan *et al.* 1998) and mitochondrial DNA sequencing (Keenan and Lavery 2001). Through these studies Keenan *et al.* (1998) have confirmed the presence of four distinct species: *S. serrata* (Forskål 1775), *S. olivacea* (Herbst 1796), *S. tranquebarica* (Fabricius 1798), and *S. paramamosian* (Estampador 1949). In confirming the genetic separation of species Keenan *et al.* (1998) also assessed morphological differences using forward stepwise discriminant function analysis. 100% discrimination between the four species was achieved using three significant discriminant functions, incorporating 16 characters. The morphological data collected during the study indicates some distinct differences between species groups, but no single character can be utilised to discriminate between all four species. The authors present a simple table identifying the key morphological characteristics useful in visual identification of each species. The characters identified provide a straightforward and sound method for rapid visual identification of the two species of *Scylla* (*S. serrata* and *S. olivacea*) that occur in Northern Territory waters.

Keenan *et al.* (1998) briefly reviews the distribution and reproductive patterns of the four *Scylla* species and suggests that it is unlikely that all four species co-exist within one region, citing differences in optimal salinity requirements for larval growth and survival for each species. The authors note that while *S. serrata* demonstrates widespread oceanic distribution throughout the Indo-West Pacific, the other three species are found in areas where salinities are less than 33 ppt, such as the South China Sea and Bay of Bengal. *S. tranquebarica* and *S. olivacea* are most common in areas where salinities fall below 31 ppt. This assessment appears sound since *S. serrata* is commonly found throughout its northern Australian distribution and *S. olivacea* demonstrates a patchy distribution confined to northern Australian estuaries affected by the tropical monsoon season.

Commercial catch monitoring and fishery independent sampling provide evidence of *S. serrata*'s dominance in the Northern Territory. *S. olivacea* can be found in small numbers (<1% of the total commercial catch) (Knuckey 1999) in northern and western areas of the Northern Territory, and has not been observed in the Gulf of Carpentaria (GOC). With approximately 75% of the Northern Territory commercial catch sourced from the GOC, the Northern Territory commercial mud crab catch is, therefore, dominated by *S. serrata* and is considered monospecific for the purposes of fisheries management (Zeroni, P. and Wood, L. 2004).

It appears that confusion surrounding the taxonomy of this species has arisen from; studies concentrated at regional scales when each of the four species spatial distribution partially overlapped; morphological similarities between species and some incorrect nomenclature. These problems have only been overcome by Keenan *et al.* (1998) revisiting the original type locality in the Red Sea and collecting additional material from around the Indo-Pacific.

## 2.3 Growth and Reproduction

*S. serrata* grows rapidly, recruiting to the fishery and reaching sexual maturity within 12 to 18 months (Knuckey 1999). Due to their hard exoskeleton, females can only mate immediately post-moult (Perrine 1978) and are able to store sperm for several months within their body cavity. They may successfully spawn a number of egg batches from the one mating encounter (Ong 1966).

*S. serrata* spawns seasonally throughout much of its range (Heasman *et al.* 1985; Quinn and Kojis 1987). Within Northern Territory waters, adult females are generally absent from the commercial catch during the wet season (Hay and Calogeras 2001; Knuckey 1999). Perrine (1978) suggests that female migration from estuarine areas is stimulated by a rapid fall in salinity. Hill (1994) examined trawl by-catch data from the Australian Northern Prawn Fishery for the period 1985 to 1987 and found 377 *Scylla serrata* had been captured in offshore waters in the Northern Territory (average depth 28.5m and distance from shore 17.9km). He noted that 87% of the mud crabs observed in the bycatch occurred during October and November; of those, 97% were female and 61.5% of these were ovigerous. He suggests that the seaward migration of adult female crabs occurs in either September or October and notes that this period is prior to the onset of the tropical monsoon season in northern Australia (December). Female crabs were not found in the prawn trawler catch after February and he concludes that the spawning season had ended.

*S. serrata* larvae poorly tolerate reduced salinity and are unsuited to estuaries (Hill 1974), yet in northern Australia this estuarine species' general distribution is primarily dominated by monsoonal rain patterns. Adult female migration to marine waters prior to spawning suggests a mechanism to maximise larval survival. Larvae remain in an oceanic planktonic state for around one month, undergoing a series of moults before returning to estuarine waters on flood tides and settling in coastal swamp habitats as juveniles (Arriola 1940; Fielder and Heasman 1978). It is from here, after further moults, that the crab eventually recruits to the fishery at around twelve months of age (Knuckey 1999), with adult crabs spending most of their 4-year lifespan within sub tidal and intertidal estuarine habitats (Hyland *et al.* 1984).

While observations of ovigerous female mud crabs have been reported from the inshore waters of the Philippines (Arriola 1940), South Africa, Mauritius (Hill 1975) and Ponape (Perrine 1978), it is likely that some of the crabs observed may not have been *S. serrata*. Considering the recent revision of the genus and the reported distribution and salinity tolerance of each species, it is probable that previous studies may unknowingly refer to species other than *S. serrata*. Close examination of the work of Arriola (1940) suggests that the species described is actually *S. tranquebarica*. Heasman (1980) reports that the extent of *S. serrata* spawning migrations in Queensland varied greatly according to hydrological conditions at the time of spawning. Hill (1974) also observed that female *S. serrata* will extrude eggs within estuaries if they cannot reach the sea, however the larvae cannot survive estuarine conditions and recruitment fails.

If the hypothesis that the migration patterns of *S. serrata* fulfil a critical reproductive requirement in finding suitable, more saline waters, then the extent of the migration should therefore relate to the proximity of such waters. Yet Hill (1994) suggests that migration patterns of *S. serrata* appear far more extensive than would be required to reach full sea-water salinity, and offers a secondary explanation of a larval dispersal mechanism enabling megalopa to reach distant habitats from their parents.

## **2.4 Distribution and Movement within Estuaries**

Stephenson and Campbell (1960) attribute the genus *Scylla*'s success to its ability to successfully invade estuarine environments, "a habitat not occupied by any other portunids or any other crab approaching its size". Hill (1979b) observed that *S. serrata* could tolerate a four-month period of low salinity (2ppt) in a South African estuarine lagoon and subsequently measured their hypersalinity tolerance at 60ppt in the laboratory. He noted that population density declined with increased distance from the mouth of the estuary studied, and that *S. serrata* was more abundant in muddy than sandy areas.

Apart from the reproductive and environmental requirements of spawning migrations, mud crabs appear to be relatively sedentary, possessing a restricted home range (Brown 1993). Hyland *et al.* (1984) tagged 6233 *S. serrata* over a 14 month period in Deception Bay, Queensland. They observed very little movement between an estuary and the connecting bay and no movement was evident between two adjacent areas separated by habitat considered unsuited to *S. serrata*. Perrine (1978) also reported similar findings of minimal movement from tagging studies in Ponape, with most crabs recaptured close to their original tagging locations. In South Africa, Hill (1975) reported that 68% of tagged crabs recovered (n = 137) had moved < 1 km, with time intervals between marking and recapture varying between 1 day and 13.5 months (mean 99 days). He concluded that

while adult mud crabs are capable of long distance movement, they remain in the same general area for extended periods. Considering the short (<4 year) lifespan of mud crabs this finding is relevant. Heasman and Fielder (1977) add that "unlike many other commercially fished species which are migratory, mud crab populations are firmly bound to particular mangrove estuaries beginning with settlement and metamorphosis of the last stage larvae (megalopa) to the first crab stage (post larvae)".

## 2.5 Diet and Predation

Feeding by decapod crustaceans is affected by environmental factors such as temperature, and physiological factors such as moult condition (Williams and Hill 1982). Hill (1980) reported that feeding activity and general movement of *S. serrata* decreased markedly at temperatures below 20°C. Northern Territory inshore sea surface water temperatures generally remain above this level, averaging between 21°C and 32°C throughout the year. (Bureau of Meteorology 2005).

Arriola (1940) describes *S. serrata* (*S. tranquebarica*) as an opportunistic scavenger and a cannibal and noted that in aquaria, crabs would capture and feed on small live fishes and shrimps using their pincers, while in the wild, large crabs prey on smaller or injured crabs as well as ingesting algae or decaying organic matter. Hill (1976), however, found that even when starved for up to 15 days, *S. serrata* was unable to catch live penaeid prawns and were only successful when the prawns became incapacitated or had died. Perrine (1978) also found that *S. serrata* were unable to catch live fish in aquaria, but did capture and devour xanthid crabs.

Hill (1976) assessed the natural diet of Australian and South African *S. serrata* and reported that 50% of the identifiable material in the foregut contained molluscan fragments, 22% contained crustacean remains, mainly grapsid crabs and the remaining 28% consisted of small amounts of plant material and debris. He concluded that *S. serrata* is predominantly a nocturnal predator of sessile or slow-moving benthic macro-invertebrates, chiefly molluscs. He also provided details of gut clearance rates of organic matter, fish bones and mollusc shell fragments at 12 hrs, 2-3 days and 5-6 days respectively. Brown (1993) acknowledges these results and highlights the need for care in interpreting gut contents analysis in dietary studies.

Hill (1976) suggests that a diet of slow moving benthic invertebrates might be a factor in limiting the distribution of *S. serrata*. In a further study Hill (1979a) identified the chief food items of *S. serrata* in South African estuaries as bivalves. He noted that estuarine bivalve mortality increases during floods due to periods of reduced salinity, and that this absence of food may reduce the carrying capacity of the system for invertebrate predators such as *S. serrata*. However it is unlikely that Hill's observations would be as relevant in the tropics where monsoonal conditions ensure bivalve populations are immersed in waters of reduced salinity for extended periods.

Arriola (1940) presents evidence of adult mud crabs ingesting plant material. Hill (1976) also found that mud crabs tend to swallow considerable amounts of indigestible material but queries whether this is part of their normal diet or merely swallowed along with other food.

Hill (1979a) and Brown (1993) document previous work describing predators of adult *S. serrata*. Hill (1979a) and Brown (1993) both note Crosnier's 1962 record of large mud

crabs forming part of the diet of sharks in Madagascar (species unknown) and South Africa (*Carcharinus leucus*). Considering the prevalence of sharks in northern Australian waters it is likely that mud crabs form some part of their diet. Hill (1979a) also reported observations of crocodiles (*Crocodylus niloticus*) feeding on *S. serrata* in South Africa. Two large predators, the saltwater crocodile (*Crocodylus porosus*) and estuarine grouper (*Epinephelus* sp greater than 1.2m) are protected in the Northern Territory and are therefore abundant across northern Australia; both have been reported to feed on mud crabs (Taylor 1979; Hay unpublished observations). Fielder and Heasman (1978) also report that turtles and large fish such as barramundi (*Lates calcarifer*) also predate on adult mud crabs in Queensland waters.

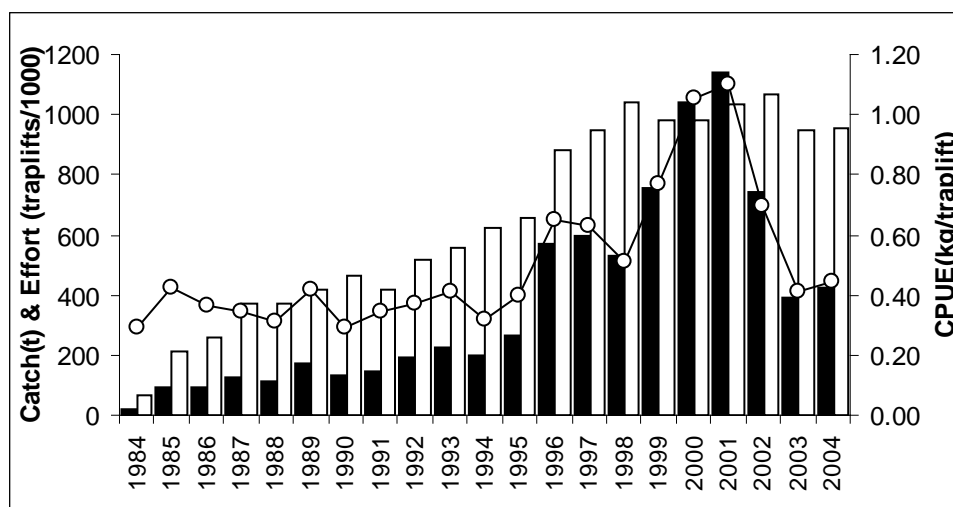
While most authors consider the impact of various predators on *Scylla* populations it appears little research has focused on the impact of large mud crabs on new recruits and small mud crabs. Moksnes *et al.* (1997) consider “that intra-year class cannibalism is a major process regulating both survival and dispersal in megalopae and juvenile blue crabs (*Callinectes sapidus*)”. While various predators impact on mud crabs, it is likely that intraspecific predation contributes considerably towards the natural mortality of this species.

## 2.6 The Fishery

The mud crab *S. serrata* fishery operates in tidal and estuarine waters of the Northern Territory and is utilised by three major stakeholder groups, the commercial, recreational and indigenous sectors. Currently the commercial sector has the greatest impact on the resource, while estimates of the recreational and indigenous sector impact are equal and increasing.

### 2.6.1 The Commercial Sector

The Northern Territory commercial *S. serrata* fishery commenced in the early 1980's. The fishery was, and remains, a low technology fishery requiring a licence, four to five metre aluminium dingy, outboard motor, crab traps, and bait. The fishery developed rapidly, with catch increasing tenfold over the first decade of operation.



**Figure 3.** The Northern Territory Mud Crab Fishery total catch in tonnes (closed histogram), effort (traplifts x 1000 -open histogram) and CPUE (kg/traplift-open circle) for 1984 to 2004.

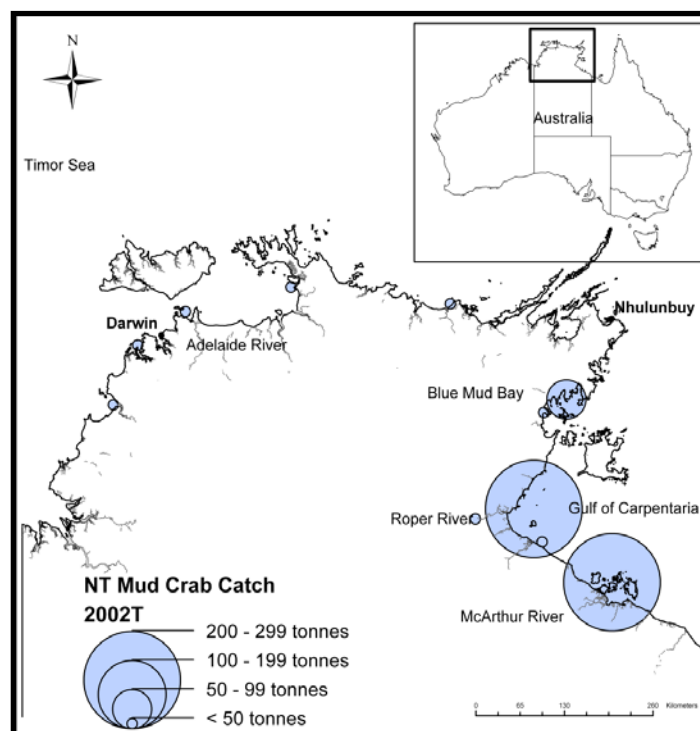


This upward trend in catch continued until 2001, reaching a peak of 1139 tonnes (t) and declining in 2003 to 393t (Figure 3).

Management strategies for Australia's mud crab fisheries are highly regulated, yet differ for each state and territory. Regulations for the commercial industry include the use of input controls, limiting fishing effort by restricting licence numbers, gear restrictions, and output controls such as size limits. Recreational fishing controls include a possession limit, size limits and gear restrictions.

The Northern Territory commercial fishery is restricted to a total of 49 fully transferable licences. Provisions exist for licensees to transfer their licence, permanently or temporarily, with the majority of licences being leased. The high value of a mud crab fishery licence and annual lease fees (approximately \$400 000 and \$30 000 respectively) provides little potential for any additional effort in this fishery with all 49 Northern Territory mud crab licences fully utilised. Each licence holder is entitled to use a maximum of 60 traps, which must comply with specified dimensions and construction materials.

The majority of commercial crabbing occurs in remote regions of the Northern Territory, mostly in the GOC (Figure 4). Crabbers construct small rudimentary bush camps in sheltered rivers that become their homes for up to ten months of the year. They generally work the creeks and rivers within a 100km radius of the base camp, often sleeping in their dinghies and returning to the base camp at periodic intervals to refuel, unload and socialise (Figure 5).



**Figure 4.** 2002 Northern Territory commercial mud crab harvest in tonnes by region.



**Figure 5.** Typical mud crab fishery base camps, located near the McArthur and Roper Rivers, Northern Territory.

### **2.6.2 Recreational and Indigenous Sectors**

Mud crabs are also harvested by the Northern Territory recreational sector and prior to 1995, the impact of recreational fishing on Northern Territory fish stocks was unknown. The 1995 Northern Territory survey of recreational fishing activity estimated recreational harvest of was 52 000 mud crabs or around 41 tonnes (Coleman 1998). The results of a second survey in 2000 estimated recreational harvest of mud crabs had increased to 82 000 crabs or around 65 tonnes (Henry and Lyle 2003), approximately 6% of the reported commercial catch (1037t) for the same year.

The mud crab is also a significant food source for northern Australian coastal aboriginal communities and the work of Henry and Lyle (2003) provides the first estimates of indigenous harvest across northern Australia. In 2000, 86 000 mud crabs were harvested by indigenous Australians in the Northern Territory, approximately equal to the Northern Territory recreational harvest for the same year. To provide some perspective, the estimated Northern Territory recreational and indigenous mud crab harvest for 2000 was 125t, which is equivalent to 12% of the commercial catch for the same year.

In the Northern Territory, recreational and indigenous fishers do not require a fishing licence. Regulations permit each angler to possess 10 crab traps and a daily possession limit of 10 crabs per person or 30 crabs per boat is applicable. Recreational fishers generally undertake crabbing in conjunction with other fishing activities in coastal and estuarine regions surrounding major population centres. As such, the commercial and recreational sectors are generally geographically isolated. Nearly all recreational crabbing is undertaken using crab traps similar to those used by commercial fishers (see Figure 23), although the occasional mud crab is caught on a line. Indigenous fishers also commonly use hooks and spears to harvest mud crabs.

All sectors are required to adhere to minimum size limits of 130mm-carapace width for males and 140mm-carapace width for females and the possession of ovigerous female crabs is prohibited.

## 2.7 Mud Crab Fishery Research

Considerable research has been undertaken on *Scylla sp.* over the past 50 years, initially focused on gaining an understanding of the biology of the species (Serene 1952; Hill 1975; Perrine 1978; Heasman 1980; Quinn and Kojis 1987 and Hill 1994). The majority of recent work has contributed to the subsequent development of laboratory and hatchery practices for aquaculture of the species.

It is widely acknowledged that high levels of localised exploitation of juvenile mud crabs for aquaculture grow out, coupled with extensive removal of critical habitat during the 1970's and 1980's, has severely impacted on South East Asian mud crab wild stocks (SEAFDEC 2000, Naylor *et al.* 2000). Consequently the majority of research in this region is now focused on aquaculture production for the reseeded of natural populations and for stocking mangrove pen culture (SEAFDEC 2002).

Within Australia, the mud crab resource has been somewhat protected. Humans lightly populate the majority of the mud crabs' geographic distribution and therefore critical habitat is mostly undisturbed. Fisheries in Australia also tend to be highly regulated and Australian mud crab aquaculture is still in early developmental stages. This aquaculture focuses on closing the life cycle rather than sourcing juveniles from the wild.

### 2.7.1 Northern Territory Fishery Assessment

Commercial fishers in the Northern Territory are legally required to provide monthly summaries of their fishing activity, providing spatial and temporal details of catch and fishing effort. Validation of these data is important so that changes in fishing practises and error in reporting can be quickly identified. In the Northern Territory, reported commercial catch and marketed catch are cross checked using airfreight quantities obtained from the airline industry, providing a measure of reporting accuracy for assessment purposes. This is due to the lack of infrastructure and the remoteness of the fishing grounds resulting in all commercial catch being road-freighted to Darwin, checked for condition, repacked and flown to interstate and international destinations. Reported effort however is far more difficult to validate. The majority of fishing activity occurs in the remote GOC making enforcement of trap restrictions difficult. Fisheries Enforcement Officers regularly report illegal-fishing activity, with possession of excess traps the most frequently reported infringement, and convictions often involve the use of double the legal number of crab traps.

While the fishery remains a low technology fishery, changes in fishing power are evident with fishers increasing the size of outboard motors on an annual basis. In 1995 most fishers in the GOC used 60-70 horsepower two-stroke outboard engines; in 2002 100-135 horsepower four-stroke outboard motors are commonly observed. Increased horsepower allows the fisher to cover more distance within the constraints of tide and also permits multiple checking of traps per tide.

An estimate of stock size is a fundamental requirement in predicting a fishery's production potential, and subsequently for developing ecologically sustainable management practices. However, no stock estimates are currently available for Australian

mud crab fisheries. Traditional stock assessment is based on the "analysis of commercial catch and effort data" as this is the primary data available in most fisheries (Hilborn and Walters 1992). A major problem in using commercial catch and effort data to estimate stock distribution and abundance is that "fishers go where the fish are". Fishing effort is normally concentrated on the highest densities of fish and attempts to assess the range, or total abundance of fish from commercial catch and effort data can be expected to be biased.

A 1996 review of the Northern Territory mud crab fishery found that catch-effort models and assessment methods based on catch-rate data could not be applied to this fishery due to the non-random fishing effort (Walters *et al.* 1997). The authors found that "it appears commercial crabbers in northern Australia operate by systematically fishing and resting local areas over the fishing season, and in this manner they maintain hyperstability in catch per unit effort". This indicates that fishery independent catch rate data is required.

During the assessment, attempts were made to fit a seasonal population dynamics model (Knuckey 1999) to monthly time series of the catch and CPUE data. The model fits imply increased recruitment levels of approximately six fold for 2000 and 2001 however it is likely that some proportion of the increased catches are due to the high levels of unreported effort and an expansion of areas fished.

The use of length-based methods for estimating growth and mortality rates of populations is not valid for this species due to the way crabs grow and the subsequent lack of discrete length cohorts in the stock (Walters *et al.* 1997). In common with all other crustaceans, mud crabs possess an exoskeleton that must be shed through a series of moults or ecdyses before growth can occur (Brown 1993), and this species exhibits high variance in individual growth rates (Arriola 1940; Ong 1966; Knuckey 1999). Many other forms of assessment of stock abundance, such as visual assessments are also inappropriate. The high turbidity of northern Australia's estuaries and inshore waters prevents visual counts (Melville-Smith 1986; Robertson 1989). Apart from during the offshore female spawning migration, trawling is an ineffective sampling method for this species due to, burrowing behaviour, fishing gear selectivity (Williams and Hill 1982; Melville-Smith 1986) and the shallow and often snag affected environment common to tropical mangrove lined waterways.

Recent declining trends in commercial mud crab catches suggest that a degree of urgency in gaining estimates of mud crab stock size is warranted. The 1996 stock assessment reported by (Walters *et al.* 1997) indicated the Northern Territory mud crab fishery was fully exploited at a time when total catch was 264t. The authors estimated that the annual exploitation rate for this fishery ranged from 70-90% of the available stock, suggesting that there was little room for further development. However, within twelve months, the total Northern Territory commercial mud crab catch doubled, reaching 595t. Commercial catches in the Northern Territory continued to increase, reaching a peak of 1139 t in 2001 before declining to 393t in 2003.

## **2.8 Estimation of Animal Abundance**

The estimation of animal abundance is an important challenge in both theoretical and applied biological sciences (Otis *et al.* 1978). As such vast amounts of scientific and statistical expertise have been devoted to improving and developing robust assessment

methods, particularly over the past 30 years. However, the study of natural aquatic populations is not easy. In some cases, visual counts can be made providing an accurate measure of absolute density for a particular species. For this to occur all animals must be visible at the time of the survey. In reality this is rarely possible, and is especially unlikely in the marine environment, as most aquatic populations are difficult to observe (Hilborn and Walters 1992). This situation is compounded in tropical estuarine environments that are characteristically turbid for most of the year. Fisheries assessments have therefore had to rely heavily on the development of alternative methods to estimate abundance of aquatic species.

The tagging of fish and other aquatic organisms is not a new approach. Mark-recapture techniques (tagging) have long provided important biological information (e.g. growth, movement and in some cases exploitation rates) for many species through tag returns. However, tagging data can also provide much more valuable information. Experiment design and analysis techniques have advanced considerably and robust estimates of population size can be achieved from mark-recapture and removal data. These advances have culminated in the development of a series of models and more recently, software that allows for temporal, behavioural and individual heterogeneity in capture and recapture rates. However, for these models to perform with minimal error, certain assumptions must be addressed, and this requires detailed experimental planning.

Seber (1986) acknowledged a long-standing problem when converting sample population estimates to density estimates: determining the actual area sampled (the "edge effect" where animals are drawn into the study site). Identifying the actual area of the study site sampled is a difficult task, as the use of baited traps may cause immigration of new animals into the study site, positively biasing the population estimate. This varies among species and is dependent on a number of factors such as the animals' home range, and a mosaic of environmental and behavioural factors. It is therefore important when using baited traps to incorporate a method to monitor animal movement into the study site.

This problem is further complicated by the fact that a species may be found in a variety of habitats. Throughout their broad geographical distribution mud crabs exhibit strong habitat preferences for soft muddy substrates found in estuarine environments such as mangrove-lined streams and mudflat foreshore areas. In order to sample these two distinct habitats, we have developed separate experimental designs that attempt to meet the associated model assumptions. For mangrove-lined streams we introduce a new design combining mark-recapture and depletion techniques, and for the foreshore mud flats we apply the trapping web design developed from Distance sampling theory (Buckland *et al.* 1993).

As these two methodologies had not previously been applied to fisheries studies, optimum application required preliminary studies. I first developed the depletion mark-recapture method used in this study in 1997 following the 1996 Northern Territory mud crab assessment recommendations (Walters *et al.* 1997).

David Anderson and Ken Burnham from Colorado State University introduced trapping web theory at a mark-recapture and distance sampling workshop held at the University of Queensland, Brisbane in June 2000. I pondered and then adapted the concept to the marine environment and the first marine application of this method was tested by Northern Territory Fisheries in October 2000. These preliminary studies conducted by

Northern Territory Fisheries Group were precursors to the research presented in this thesis.

### 2.8.1 Mark-Recapture and Depletion Theory

Mangrove-lined streams provide a natural boundary to immigration/emigration simply by having defined edges to the habitat area (i.e. banks) with only the upstream and downstream portion open to influx or escape. This project has focused on developing an experimental design that combines mark-recapture and depletion techniques. This novel design provides a method for assessing the assumption of closure by permitting a measure of the movement of tagged animals into the study site.

Mark-recapture methodology is very comprehensive (Pollock 1991). A simple example of a mark-recapture experiment follows, where a sample of  $n_1$  animals is caught, marked and released. Later a second sample of  $n_2$  animals is captured, of which  $m_2$  are recaptures from the first survey. An intuitive estimator of the population size ( $N$ ) can be achieved based on the assumption that the ratio of marked to total animals in the second sample should reflect the same ratio in the population (Pollock 1991) and  $N$  is thus estimated as  $\hat{N} = n_1 n_2 / m_2$ . The precision of this estimate can be improved by adding further recapture occasions, provided the population remains closed (i.e. no immigration/emigration). Choosing the sampling duration for such a study is therefore an important factor in the study's success. Otis *et al.* (1978) recommend that a closed population model requires 5 to 10 sampling periods with a minimum average capture probability of 10 % per period for sound results. Also worth noting for many of the density estimation models, sample sizes in the range of 80-100 individuals have been recommended by various authors (Otis *et al.* 1978; Burnham *et al.* 1980; Buckland *et al.* 1993).

Five general assumptions are recommended for estimating animal abundance using mark-recapture methods;

1. The population is closed (no immigration/emigration).
2. Animals do not lose tags during the experiment.
3. All marks/animals are correctly noted and recorded at each trapping occasion.
4. Each animal has a constant and equal probability of capture on each trapping occasion.
5. All marked animals mix with the unmarked animals and have equivalent recapture probabilities.

The assumption of population closure is perhaps one of the most important. A closed population is one where the population is constant for the duration of the experiment (i.e. no births, deaths, immigration or emigration). This is a strong assumption and in reality has proved difficult to achieve (Otis *et al.* 1978; White *et al.* 1982). Pollock (1982) and Seber (1992) expand this further by suggesting that an open population, where migration, births and deaths occur, can sometimes be treated as closed if the study period is short enough. This is the approach we have taken, combined with movement studies to test this assumption.

The two assumptions relating to tag application and recording are often overlooked in large tagging programs. It is important that all personnel involved in conducting the

tagging and recording of data are trained in applying tags and recording data. As such the Northern Territory Fisheries Group has prepared a training manual for all staff employed on the project.

Otis *et al.* (1978) considers the wide recognition that assumption 4 commonly fails and that accurate population estimation usually requires models that provide for unequal probabilities of capture. Schwarz and Seber (1999) acknowledge the extensive research directed at both examining the effects of departure from assumption 4 on estimates and the modification of various models to allow for such departures. White *et al.* (1982), Pollock (1991), Chao (1987) and Chao *et al.* (1992) all provide estimators that do not assume equal catchability.

The final assumption requires that marked animals mix freely with the unmarked population on return and do not become “trap happy” or cluster around the trap. A population that meets this assumption is one where the ratio of marked to unmarked animals remains unchanged.

The removal or depletion method is particularly suitable for estimating fish and aquatic invertebrate populations (Carle and Strub 1978), especially when it is possible to fish a population intensively over a short period of time (Hilborn and Walters 1992). In a removal study animals are trapped and removed from the population rather than marked and released. On the first and all subsequent occasions all captured animals are removed. The general concept of this form of assessment is to examine how the deliberate removal of the species influences the relative abundance of the remaining population in the depletion study area (Hilborn and Walters 1992).

The two general assumptions for removal studies are equivalent to assumptions 1 and 3 for mark recapture methods.

### **2.8.2 Trapping Web Theory**

Open foreshore mud flat areas provide an interesting challenge in estimating animal abundance. The lack of boundaries in open forests and terrestrial systems are recognised as an important factor that may positively bias estimates. This is due to the edge effect, where animals from outside the study site may be drawn into the sampling area. In attempting to deal with edge effect Anderson (1983) introduced the trapping web study design. This method also uses closed population capture-recapture data and provides additional tools to account for unknown biases caused by the edge effect. Additionally for a trapping web to work the experiment duration must be short to permit the assumption of a closed population and thereby minimising the opportunity for large-scale movement, migration and recruitment to positively bias the estimate. It is also important that as large a number of traps be used as is practical, to ensure a high probability of capture at the centre of the web.

Distance sampling is a widely-used group of closely related methods using point or line transect data to estimating animal abundance or density (Buckland *et al.* 1993, Thomas 2002, Borchers *et al.* 2002). In essence, upon completion of a survey,  $n$  animals have been detected and their associated distances from the observer are recorded (Buckland *et al.* 1993). There is a marked tendency for detection ability to decrease with increasing distance from the observation point and the analysis incorporates a correction factor based

on the distance data to correct for undetected animals (Buckland *et al.* 1993). Sampling is conducted over a number of consecutive occasions until no new animals are caught near the centre (Buckland *et al.* 1993). Typically the experiment duration is between three and eight sampling occasions (Buckland *et al.* 1993).

Distance sampling theory centres on the measurement or estimation of the detection probability  $g(x)$ , where the probability of detecting an animal decreases as the distance between the observer and the animal increases. Several flexible models of  $g(x)$  can be implemented using this method of analysis. The four available key functions are the uniform, half-normal, hazard-rate, and negative exponential models (see Chapter 2.8.2 Buckland *et al.* 1993). A 'series expansion' is used to adjust the fit of the model of distance data. The available series expansions are cosine, simple polynomials, and hermite polynomials. This fitted function allows the estimation of the proportion of objects missed by the survey.

The recommended analysis strategy is to select a few models for  $g(x)$  that demonstrates the key properties of model robustness, shape criterion, and efficiency.

Anderson *et al.* (1983) modified this sampling theory and introduced the radial trapping web design. The authors propose that this design eliminates problems associated with most mark-recapture studies, such as accurately measuring effective trapping area and estimating capture probabilities. The method has been widely adopted with various studies examining variations of the method (Link and Barker 1994; Wilson and Anderson 1985; Parmenter *et al.* 2003). While animals are marked and released only the first capture dataset is required for analyses and subsequent recapture information is used to identify the point in time when all individuals have been marked.

Seber (1986) conveys the benefits of the Anderson *et al.* (1983) trapping web design and the nonparametric distance sampling technique of Burnham *et al.* (1980) as;

- the avoidance of a rectangular trapping grid (i.e. accounts for the edge effect);
- trap density near the centre of the web is very high to allow all animals at the centre of the web to be captured with certainty; and
- rigorous methods of analyses are available.

The trapping web design consists of a number of lines of equal length, radiating from a central randomly chosen point. At equal distances along each line are a consistent number of traps. Buckland *et al.* (1993) suggest a minimum use of at least eight lines of traps. They state that 'the trapping web was envisioned for use with animals that have some form of home range or territory' and hence 'trap spacing along each line should be determined with respect to the home range of the animal under study'. The spacing should be such that there are 8-12 traps (or more) per home range in the centre of the web (Anderson *et al.* 1983).

As with all abundance estimation methods a number of assumptions underlie the robustness of these analyses. Buckland *et al.* (1993) state three assumptions are essential for reliable estimation of density from distance sampling. Ordered from most to least critical, these are:



(1) All animals at the centre of the web are captured at least once during the sampling occasions. That is, trapping continues until evidence exists that no new animals are being caught near the centre of the web.

This assumption 'is critical but can be monitored by examining the number of new individuals trapped near the web centre over the trapping occasions'. Alternatively, if at the centre of the web most animals that have been marked and released have subsequently been recaptured then one might conclude that sufficient trapping occasions have been carried out.

(2) Objects are detected at their initial location, prior to any movement in response to the observer.

(3) Distances from the centre of the web to each trap are measured accurately.

A study by Anderson & Southwell (1995) evaluated the skill levels required to effectively analyse distance data. The authors offered data sets to recently trained students and experts for analysis and examined the skill levels required to competently analyse distance data, using the programme Distance. The data used in the study was collected under a suitable design, and when assumptions were relatively valid. The results were surprising with little difference in the analyses conclusions from either group. Therefore, Anderson and Southwell concluded that researchers with basic training could perform at a similar level as experts when undertaking distance sampling analysis.

A further study conducted by Cassey & McArdle (1999) examined Distance sampling performance using a range of distributions, densities and detection abilities through computer simulation. They found that, aside from violation of the assumptions of distance sampling, analyses consistently reported accurate density estimates.

## **3.0 HABITAT**

### **3.1 Introduction**

The importance of coastal wetland habitats to fisheries and fishery production are well documented (Robertson and Duke 1987, Ronnback 1999, Manson *et al.* 2001, Melville and Connolly 2005). Mud crabs due to their largely inshore benthic life history are highly reliant on these inshore habitats including mangroves and foreshore flats for settlement, protection, reproduction, and predation/feeding (Walton *et al.* 2006).

A recent study by Manson *et al.* (2005) analysed the links between coastal fisheries production and mangrove extent by investigating patterns of coastal environments, mangroves and commercial fisheries in Queensland. Results of the research suggested that mangrove characteristics, particularly perimeter, were the dominant parameters in explaining the variation in local CPUE for mud crabs, which are known to inhabit mangroves as juveniles.

The habitat datasets required for this study were compiled by the Queensland Department of Primary Industry and Fisheries (QDPI&F) Assessment and Monitoring Unit (in Hay *et al.* 2005). The mapping was completed using remote sensing techniques, which including classification of images from Landsat 5 & 7 Thematic Mapper (TM) and Enhanced Thematic Mapper (ETM), aerial photography and ground truthing (Danaher 1995; de Vries *et al.* 2002).

### **3.2 Methods**

Additional GIS analysis of the wetland coverage's were carried out using commercially available software ARCGIS (Version 8). The Northern Territory fishery catch and effort data (Appendix 1) was overlaid with the Northern Territory's 60 nm reference fishing grid. Additional spatial datasets were acquired from the TOPO-250K Series 2 GEODATA from Geoscience Australia (AUSLIG 1994). These hydrography datasets included major rivers and drainage for the Northern Territory were projected, merged and overlayed with the Northern Territory wetlands datasets, and then intersected with the Northern Territory fishing grid datasets. The area of mangrove-lined tidal rivers and streams that intersected with wetland data was calculated using standard ARCGIS area and length scripts. This provided estimates of the length of mangrove lined waterways, and area of mud flat foreshore for each fishing grid.

### **3.3 Results**

Estimates of habitat area per vegetation class were calculated for each fishing grid (Table 1). Each grid covers a 60 nautical mile square or 111.12 km<sup>2</sup>.

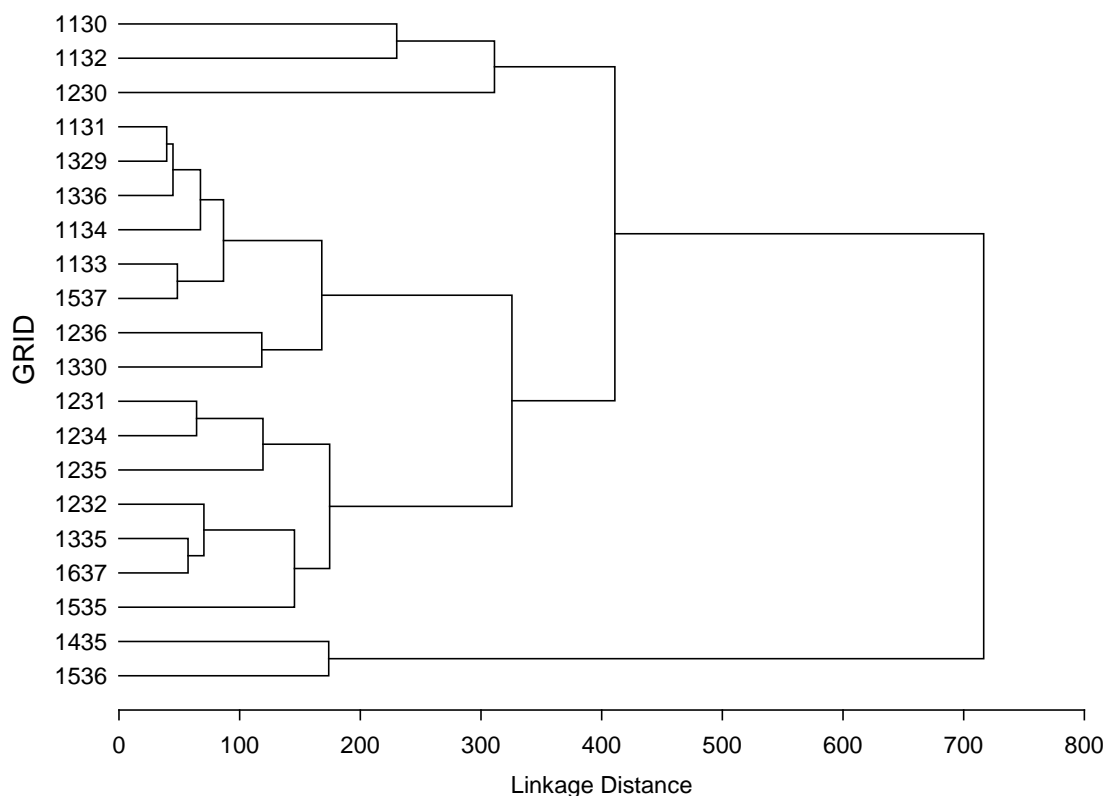
**Table 1.** Estimated area (km<sup>2</sup>) Northern Territory coastal habitat/ vegetation class for each Northern Territory fishing grid (see Appendix 1) in km<sup>2</sup>

Vegetation/Habitat Class	Area of habitat class (km <sup>2</sup> ) for each Northern Territory Fishing Grid															
	1032	1130	1131	1132	1133	1134	1135	1136	1230	1231	1232	1234	1235	1236	1329	1330
Closed Avicennia	0.12	24.36	51.11	26.60	21.56	4.96		0.08	17.39	103.90	42.09	67.15	7.81	20.11	6.08	26.27
Closed Ceriops	1.77	76.83	36.76	112.10	30.45	7.33	4.72	0.68	199.67	53.09	0.16	19.37	71.88	57.36	6.54	1.21
Closed Mixed	0.41	89.55	33.56	21.64	10.41	3.78	5.28	0.14	139.51	75.25	45.01	75.84	129.13	55.17	12.85	8.99
Closed Rhizophora	0.06	287.57	132.73	156.14	27.24	2.91	7.18	3.78	33.05	41.47	18.89	35.04	36.53	50.30	1.81	2.91
Closed Avicennia/Ceriops	0.01	35.34	21.82	13.23	2.10				0.52	13.68	4.55	1.74	2.31	2.76	0.29	
Closed Sonneratia		2.24	0.28	1.68		0.07			14.28	2.20	6.00	0.46	4.87	3.65	1.24	0.90
Open Avicennia		1.23	2.34	2.96	1.12	0.09			4.95	8.94	8.53	2.48	0.24	4.13	1.21	13.15
Open Avicennia/Ceriops			0.06	3.06												
Open Ceriops		0.44	0.00						0.05			0.03		0.12		
Open Sonneratia		0.77	0.33	4.32		0.01			0.65		0.88	0.42	0.41	0.40		
Saline Grassland		0.50	0.00						0.18	4.56			0.16	0.78		
Sedgeland			0.00						1.56							
saltpan	0.29	44.73	99.36	215.98	64.56	5.60	2.37	0.32	64.17	243.76	231.05	208.44	224.70	114.66	11.36	41.11
Samphire-dominated saltpan		1.05	2.37	2.92	3.51	0.20	1.24		1.87	9.80	9.40	8.29	16.84	2.79	0.71	7.39
water and terrestrial	0.09	7.78	8.04	65.77	6.99	0.61	0.29	0.06	10.11	35.61	46.76	36.80	19.96	5.75	0.43	4.96
<b>Foreshore mud flats</b>	<b>3.04</b>	<b>142.19</b>	<b>63.05</b>	<b>148.03</b>	<b>55.24</b>	<b>13.89</b>	<b>31.79</b>	<b>5.57</b>	<b>254.98</b>	<b>98.41</b>	<b>15.71</b>	<b>110.62</b>	<b>65.45</b>	<b>107.23</b>	<b>61.76</b>	<b>137.37</b>
<b>Mangrove lined stream</b>	<b>0.2</b>	<b>117.9</b>	<b>41.5</b>	<b>76.1</b>	<b>21.86</b>	<b>0.55</b>	<b>0.44</b>	<b>2.59</b>	<b>24.93</b>	<b>37.65</b>	<b>162.7</b>	<b>47.03</b>	<b>47.43</b>	<b>24.7</b>	<b>6.96</b>	<b>52.14</b>
<b>CLASS</b>	<b>1335</b>	<b>1336</b>	<b>1428</b>	<b>1429</b>	<b>1430</b>	<b>1435</b>	<b>1436</b>	<b>1528</b>	<b>1529</b>	<b>1530</b>	<b>1535</b>	<b>1536</b>	<b>1537</b>	<b>1636</b>	<b>1637</b>	<b>1638</b>
Closed Avicennia	52.00	29.07	0.58	83.87	0.07	36.45	0.78	0.08	42.58	0.23	14.10	65.91	1.87		9.32	0.29
Closed Ceriops	5.96	3.65		18.73		7.92	0.40				4.54	1.49			0.22	
Closed Mixed	3.82	1.38		113.09	1.37	50.89	0.16		11.62	0.08	18.05	48.76	2.93		5.25	0.20
Closed Rhizophora	12.04	21.76		23.23		6.84	0.75		7.27		4.18	16.66	0.68		1.95	
Closed Avicennia/Ceriops	6.72	7.06		0.02		6.77					10.77	7.47	0.48		2.40	
Closed Sonneratia				3.89												
Open Avicennia	3.38	3.59	1.35	67.95	0.04	4.95	0.09	0.10	68.83		0.60	5.12	0.01		0.50	
Open Avicennia/Ceriops																
Open Ceriops	0.72						0.05								0.17	
Open Sonneratia																
Saline Grassland						0.78										
Sedgeland	23.09	18.10														
saltpan	277.86	26.58	147.78	1207.9	6.44	694.23	0.76	51.28	648.01	1.22	309.12	672.67	79.96	18.73	252.33	6.36
Samphire-dominated saltpan	2.68	0.31		1.74		51.56			4.30			5.84		0.02	0.58	
water and terrestrial	3.01	0.26	7.91	39.60	0.08	50.68	0.04	1.80	36.65		10.88	26.04	5.50	1.13	21.35	0.40
<b>Foreshore mud flats</b>	<b>0.11</b>	<b>66.54</b>	<b>49.25</b>	<b>275.14</b>		<b>87.08</b>	<b>5.24</b>		<b>8.25</b>	<b>3.45</b>	<b>130.13</b>	<b>250.39</b>	<b>51.75</b>		<b>10.20</b>	
<b>Mangrove lined stream</b>	<b>10.96</b>	<b>9.35</b>		<b>170.15</b>		<b>52.78</b>	<b>7.78</b>		<b>253.41</b>		<b>50.3</b>	<b>36.11</b>	<b>5.49</b>		<b>35.41</b>	

### 3.3.1 Cluster Analysis of Grids Based on Habitat Composition

Similarities in the habitat composition of grids were analysed using a clustering algorithm. Variables included mud flats, closed *Avicennia* sp., closed *Avicennia* sp. and *Ceriops* sp., closed *Ceriops* sp., closed mixed, closed *Rhizophora* sp., closed *Sonneratia* sp., open *Avicennia* sp., open *Avicennia* sp. and *Ceriops* sp., open *Ceriops* sp., open *Sonneratia* sp., saline grassland, salt pan, samphire dominated salt pan and sedge.

The analyses were performed on the area of each habitat type (in square kilometres) present in each Northern Territory fishing grid (See Appendix 1). The complete linkage procedure (furthest neighbour) was used to determine naturally distinct "clumps" (Figure 6). All data used was measured on the same scale so joining measures were based on Euclidean distances.



**Figure 6.** Cluster analyses of habitat associations between Northern Territory fishing grids.

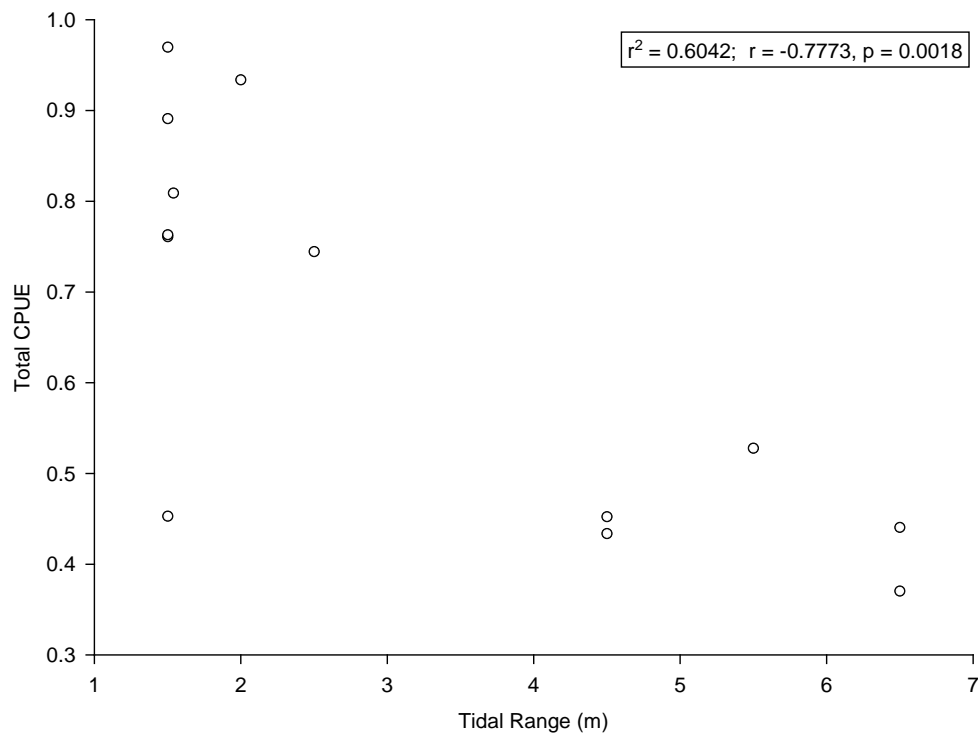
Two grids that have historically recorded the greatest Northern Territory fishing activity and catch demonstrate significant difference from any other grid-habitat type (grids 1536 McArthur/Wearyan River and 1435 Roper River). Grids 1130 (Bathurst Island), 1132 (Cobourg Peninsula) and 1230 (Bynoe Darwin) are also grouped, identified with similar characteristics but differing significantly from other areas. Two other major groupings are evident grids 1131 to 1330 and 1231 to 1535.

Following the findings of Manson *et al.* (2005) an examination of the linear correlation of habitat variables and total fishery CPUE (1995 to 2003) was undertaken (Table 2). A Plot of the most significant linear correlation is provided for total CPUE and tidal range (Figure 7).

**Table 2.** Linear correlations of total CPUE with habitat variables (all significant correlations marked in bold, where  $p < 0.05$ ).

Variable – Area Vs CPUE	r	p-value	% tot mangrove area
1. mud flats	-0.2623	0.3870	
<b>2. tidal range</b>	<b>-0.7773</b>	<b>0.0020</b>	
3. closed.avicennia	-0.2733	0.3660	23.27
4. closed.avicennia cerriops	0.1238	0.6870	4.0
5. closed.cerriops	-0.4945	0.0860	23.39
6. closed mixed	-0.5448	0.0540	24.93
7. closed rhizophora	-0.3651	0.2200	19.19
<b>8. closed sonneratia</b>	<b>-0.5802</b>	<b>0.0380</b>	1.5
<b>9. open.avicennia</b>	<b>-0.7384</b>	<b>0.0040</b>	3.18
10. open.avicennia .cerriops	-0.1805	0.5550	0.16
11. open.cerriops	0.1247	0.6850	0.06
12. open.sonneratia	-0.3175	0.2900	0.33
<b>13. mangrove total</b>	<b>-0.6136</b>	<b>0.0260</b>	100
14. saline grassland	-0.3270	0.2750	
15. saltpan	0.2347	0.4400	
16. samphire saltpan	-0.0573	0.8530	
17. sedge	0.3821	0.1980	
18. water terrestrial	-0.2898	0.3370	

Note – grids were selected for only those that had consistent (more than 4 years) CPUE data. The grids used were: 1132, 1230, 1231, 1232, 1236, 1330, 1335, 1336, 1435, 1535, 1536, 1537 and 1637. (Excluded grids: 1130, 1131, 1133, 1134, 1234, 1235, 1329 and 1636)



**Figure 7.** Linear correlation of total CPUE against tidal range in (m)  $r^2 = 0.6$  for each Northern Territory fishing grid.

### 3.4 Discussion

Linear regression of total fishery CPUE and area of habitat class provided some unexpected results. Initial analysis revealed that tidal movement may have some impact on catch rate with fishing grids with smaller tidal movement recording highest total CPUE. This observation deserves further investigation, this result may indicate a substrate preference or perhaps recruitment success is hindered by large tidal movement. An alternative suggestion could be that tidal movement affects catchability independent of density. Crabs may only move (and thus encounter a trap) when the strength of the tide is below a certain value. In regions of smaller tides there would be a greater time when the tide was below the threshold and thus crabs would have greater time to find a trap.

The assumption of strong association of crab abundance with mangrove area was not upheld with the highest catch rates occurring in fishing grids with the least mangrove area.

These CPUE-mangrove area results support the findings of Manson *et al.* (2005). The authors examined the links between coastal fisheries production and mangrove extent by investigating patterns of coastal environments, mangroves and commercial fisheries in Queensland. While area and perimeter of mangroves was significant for some inshore species, only perimeter was significant in explaining variations in the Queensland mud crab CPUE data.

In light of these results I reviewed the plan to use the total area of mangrove as a measure of habitat. In considering the use of mangroves by mud crabs it is likely that the interface between the mangroves and the estuarine environment provides the most value to this species, not the full extent or area of the mangrove forest. Further datasets detailing the hydrography of the Northern Territory were obtained and I re-analysed the datasets overlaying the wetlands coverage's over waterways and calculating the extent of mangrove lined waterways.

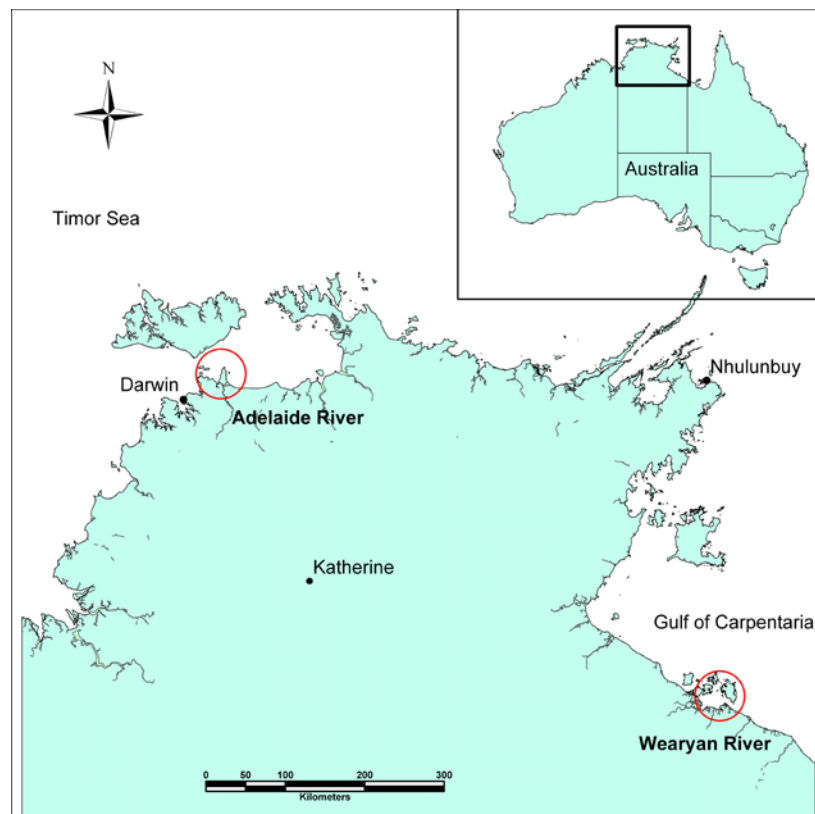
Finally in attempting to interpret these results it is important to re-iterate previous findings by various researchers that CPUE data does not accurately represent mud crab abundance (Robertson 1989; Walters *et al.* 1997). Additional cause may also be attributed to inconsistent fishery reporting (under reporting of fishing effort) and also remote conditions common to the Northern Territory; poor road access, lack of infrastructure and seasonal weather patterns inhibit consistent fishing activity in a large number of fishing grids.

Further studies investigating the linkages of habitat and species abundance may yet reveal fine scale linkages not observed in this initial study.

## 4. STUDY SITES

### 4.1 Description of Study Sites

Two locations were selected for the study of mud crab abundance in the Northern Territory (Figure 8). An area adjacent to the Adelaide River, around 60 km east of Darwin was selected (Figure 9) and the second site was the Salt Creek Estuary located adjacent to the Wearyan River in the southern GOC (also the lunar study site see Figure 10.)



**Figure 8.** The two Northern Territory study sites located adjacent to the Adelaide and Wearyan Rivers.

The characteristics of the various systems found at the two sites are presented in Table 3. Personal observations of commercial fishing activity at both study sites demonstrate differences in crab behaviour or habitat preference. Crabs from the Adelaide region tended to be caught from within mangrove-lined stream and river systems while the majority of fishing activity for mud crabs in the GOC occurred on the foreshore flats. These differences between sites can most likely be attributed to the tidal range at each site. Tides up to 7 metres (m) regularly inundated the Adelaide study site while tides up to 2.3 m occur in the southern GOC. Hill (1975) reports that crabs prefer sheltered habitat and it is likely that, in areas subject to greater tidal range, mud crabs seek shelter in creeks or rivers particularly during spring tides, while in areas of lesser tidal range crabs burrow and remain within the mud flat habitat.

**Table 3.** Comparative characteristics of the Adelaide River, Salt Creek Estuary and Fat Fellows Creek. (Source: modified from National Land and Water Resources Audit. Estuaries Assessment 2000. <http://www.ozestuaries.org/oracle/ozestuaries/> and CSIRO Simple Estuarine Response model II. <http://www.per.marine.csiro.au/serm2>)

Character	Adelaide River	Salt Creek/Twin Sisters Estuary	Fat Fellows Creek
Longitude	131.214	136.901	136.992
Latitude	-12.204	-15.911	-15.868
Condition	largely unmodified	near pristine	near pristine
Classification	tide dominated	tide dominated	tide dominated
Sub-classification	tide-dominated estuary	tide-dominated delta	tidal flat/creek
Water area (km <sup>2</sup> )	53.92	12.33	8.81
Entrance width (km)	3.85	3.56	1.29
Perimeter (km)	254.83	55.17	79.63
Maximum length (km)	68.70	10.18	16.35
Catchment area (km <sup>2</sup> )	7216	928	156
Tidal range (m)	6.5	2.30	2.30
River Flow GL/yr	1468.817	43.653	7.338
Tidal period	semi diurnal	diurnal	diurnal
IMCRA class	Beagle-Van Diemen	Pellw Gulf Coastal	Pellw Gulf Coast

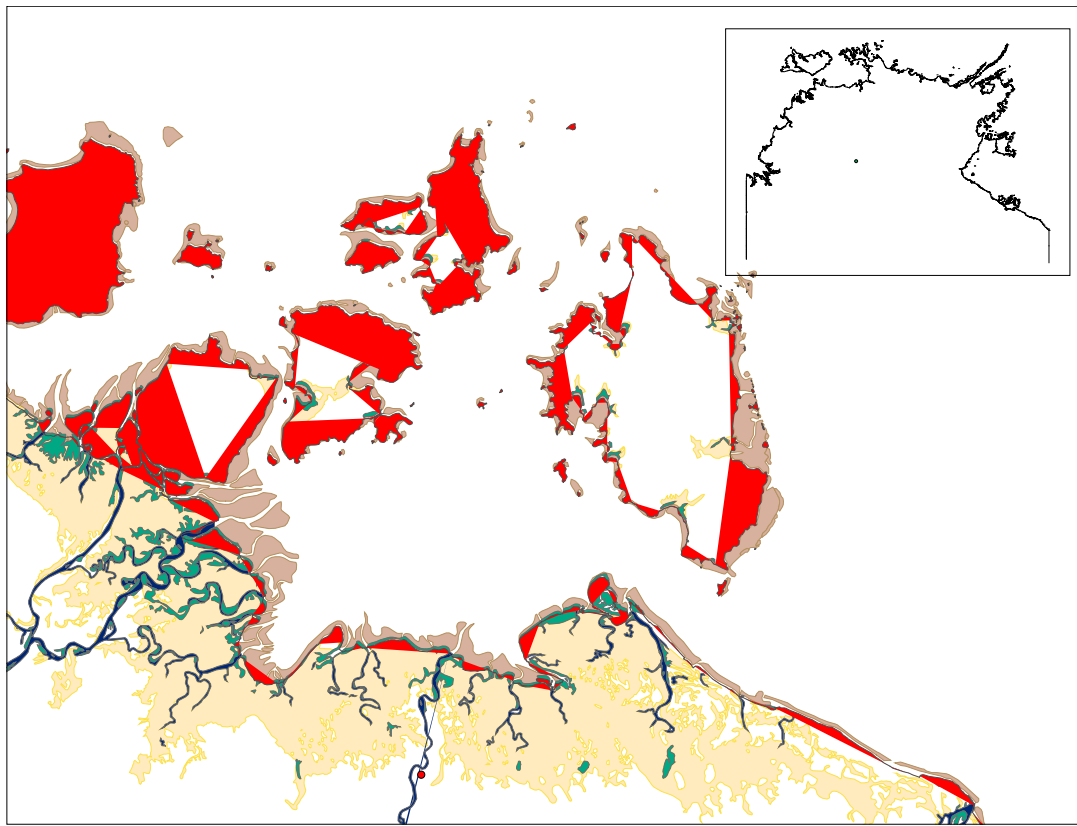


**Figure 9.** Adelaide River depletion and trapping web study sites.

The site chosen for the depletion study at Adelaide River was an unnamed mangrove-lined creek with an approximate average spring high-tide width of 30m (Figure 9). The site chosen for the GOC region depletion study was Twin Sisters Creek (see Figure 10) with an approximate average spring high tide width of 70 m. Both locations demonstrate historical



commercial and recreational mud crabbing activity and have typical areas of the two identified habitat types.



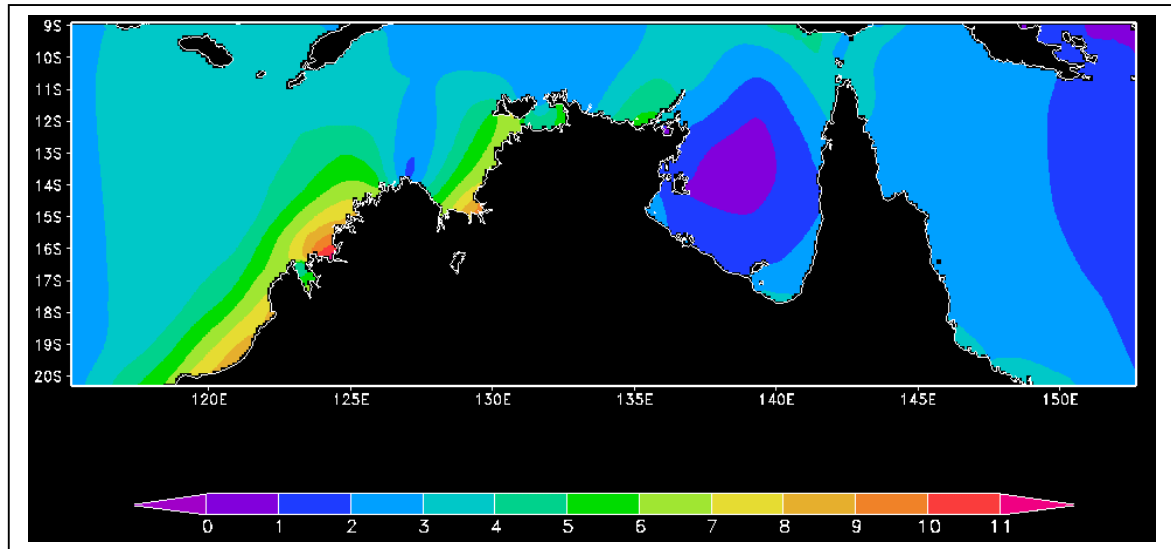
**Figure 10.** Location of the GOC lunar study sites at Salt Creek/Twin Sisters and Fat Fellows Creek.

The GOC is a very large (~370 000 km<sup>2</sup>), shallow (<70 m) body of water situated between northern Queensland and the Northern Territory. In general, the GOC floor slopes very gradually from the coastline, with sinuous gutters of deeper water radiating seaward from the mouth of every major river. The GOC substrate consists primarily of mud and sand (Staunton-Smith *et al.* 1999), with some of the muddiest areas occurring in the shallow bays and near areas of river discharge (Somers and Long 1994). Intertidal mudflats up to 5 km wide occur throughout the GOC and are most common in the south-west (Connors *et al.* 1996).

Amongst Australian drainage systems the GOC Rivers rate highly in terms of average annual discharge. They are, however, highly dependent on the austral summer monsoonal rains (Munro 1972). Median and mean annual rainfall at Boorooloola (nearest rain gauge to the lunar study site) are 731 mm and 790 mm respectively (Connors *et al.* 1996).

Sea grass beds up to 5 km wide occur throughout the south-western region of the GOC. The sea grass beds are dominated by *Halodule uninervis* and *Halophila ovalis* (Connors *et al.* 1996). Twenty-six species of mangroves are also known to occur in the coastal fringe of this region (Connors *et al.* 1996).

The GOC hydrology is dominated by an internal circulation largely isolated from major oceanic currents. Clockwise circulation of coastal waters is evident for most of the year (Church and Forbes 1983). However, this may be reversed in the wet season, particularly during intense north-westerly monsoonal episodes (IMCRA 1998).



**Figure 11.** Tidal range in metres (Model ntf\_au) for northern Australia. Source: ©Commonwealth of Australia 2004, Bureau of Meteorology, National Tidal Centre.

Tidal range across northern Australia, generally, increases in magnitude in a westerly direction (Figure 11). The GOC tidal range can be described as micro-tidal; ranging from 1-2 m offshore and 2-3 m inshore IMCRA (1998). The GOC tides are diurnal and the cycle occupies approximately one lunar cycle. The cycle is divided by the neaps into two fortnightly periods. Tides inundate the mudflats regularly, while coastal salt pans are inundated irregularly (spring tides, cyclones) (Conners *et al.* 1996). The sea surface temperature of the GOC is reported to vary by approximately 8 °C within years (IMCRA, 1998).

## 5. PRELIMINARY STUDIES

This chapter details the preliminary studies that were required to examine the various uncertainties and assumptions of the methods employed in this thesis.

### 5.1 Catchability

Estimation of crab abundance based on a feeding response, requires an understanding of all the processes involved in the capture process. Two key factors influence the capture of crustaceans in baited traps: the relationship between animal abundance and the effectiveness of the fishing gear (commonly termed catchability), and the mosaic of physiological, environmental and behavioural interactions that affect the species of interest.

The catch of a baited trap is the result of a series of interactions between the animal's attraction to the bait, the environment, and the trap (Bennett 1974 in Robertson 1989). "Trap catches are, therefore, affected by many factors, both biotic (e.g. physiological and reproductive condition of the target species, feeding rhythms, intraspecific attraction or competition) and abiotic (e.g. temperature, tidal cycle, shape and size of the entrance to the traps)" (Robertson 1989).

Williams and Hill (1982) identify two important factors affecting the feeding activity of decapod crustaceans; the physiological state of the animal (i.e. the moult stage) and, the environmental conditions, such as water temperature. Activity and appetite of decapods (and numerous other species) often increases with temperature and bait attractiveness also increases due to the rate of diffusion of bait molecules increasing with temperature (Morrissy 1975).

Traps provide a passive method of capture where capture success is acutely linked to the behaviour of the target species (Fogarty and Addison 1997) and the ability of the gear to retain captured animals.

Arreguin-Sanchez (1996) defines catchability as a measure of the interaction between the resource abundance and the fishing effort. The author identifies the two key parameters of interest as "gaining a measure of the fishing gear efficiency (selectivity) and the relationship between population size and fishing effort".

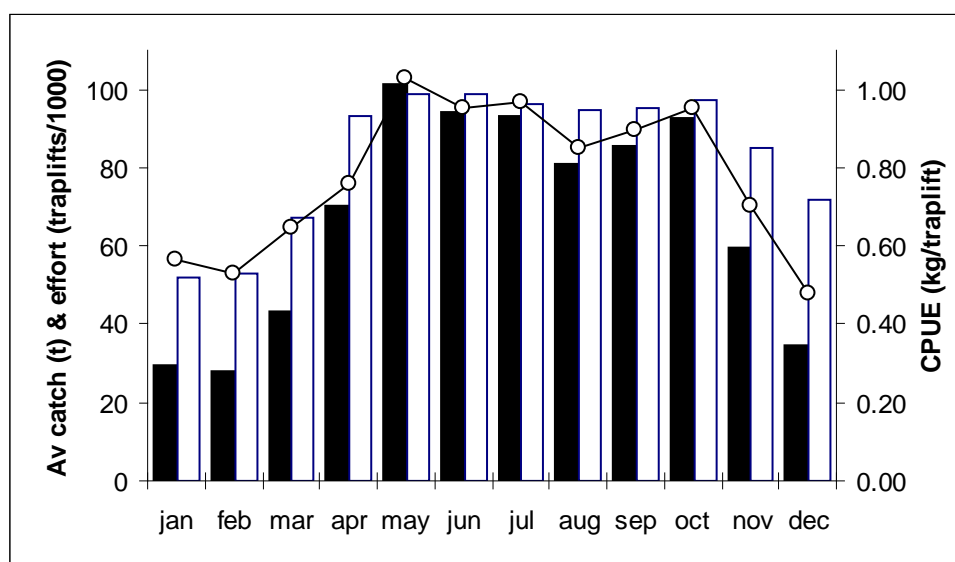
It is not unusual for animals to demonstrate some form of natural response or learned behaviour when confronted with a baited trap. Some animals become "trap-happy" while others will not be observed again and are termed "trap-shy". Summerlin and Wolfe (1973) in studying the patterns of trapping behaviour of the cotton rat, *Sigmodon hispidus* found a distinct pattern of dominant animals inhibiting capture of subordinate animals. This behaviour diminished as dominants were removed. Miller (1990) suggests that aggressive behaviour of large lobsters and crabs may also deter small animals from entering a trap. He noted frequent agnostic encounters between crabs (*Cancer productus*, *C. irroratus* and *Hyas araneus*) inside and outside traps and suggests that this behaviour may reduce entry of more crabs to a trap. The author also notes laboratory studies on several species of decapods demonstrating that larger conspecifics usually win agonistic encounters and it is only when the size differential is at its greatest that conspecifics appear to ignore one another (Schrivener 1971 in Miller 1990). Chittleborough (1974) studied the associated trapping behaviour of Western rock lobster (*Panulirus longipes cygnus*) and found that while younger individuals were abundant in the

study site, competition for food resulted in a dominance of the larger juveniles preventing the smaller lobsters from entering baited traps.

Robertson (1989) suggests that large male *S. serrata* display intimidatory behaviour which may reduce entry to a trap and encourage escape. Observations from the field program described in this thesis suggest that large *S. serrata* initially dominate trap catches and it is only when large animals are removed that smaller individuals become more abundant in the catch. This is similar to the observations of Frusher and Hoenig (2001) who found that removal of large southern rock lobsters (*Jasus edwardsii*) from a Tasmanian population by trapping resulted in increased catchability of smaller lobsters, demonstrating a size related dominance hierarchy affecting trap selectivity.

### 5.1.1 Commercial Catch Variation

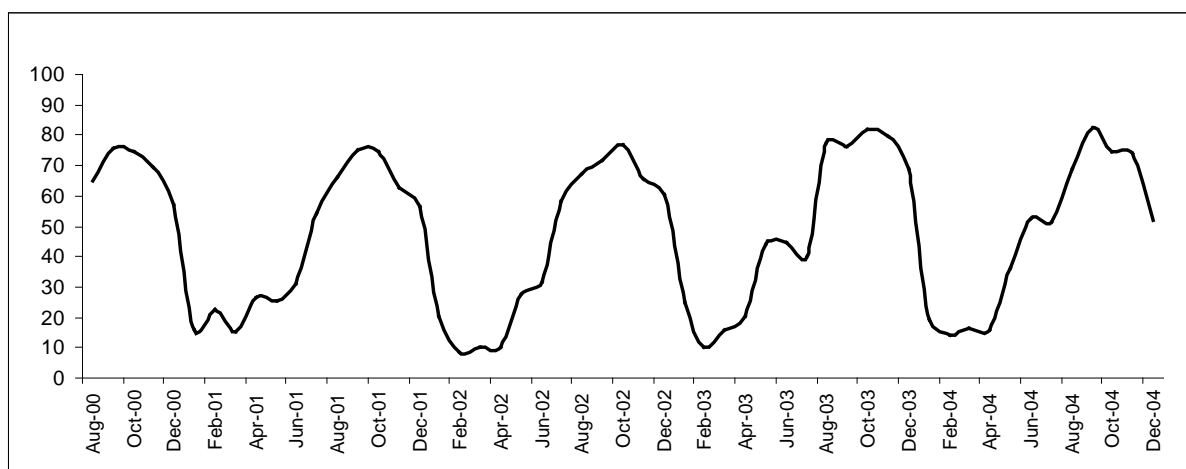
Commercial catch and effort data for the Northern Territory mud crab fishery has been collected since 1983 (Figure 3). High levels of variation in catch are apparent and season factors appear to influence catch and catch rates. Mean catch and effort data for the period 1999 to 2003 demonstrates a seasonal trend with 67% the total catch harvested during the six month dry season period (May-October) (Figure 12). This seasonal variation may be partially explained by key biological and environmental factors, such as reproduction, growth and the onset of the monsoon. Variation in levels of fishing effort is also apparent with an average of 58% of total effort applied during the dry season. Unfortunately due to remoteness of the fishing grounds, reported fishing effort is difficult to validate and use of excess traps is the most common offence in remote regions of the fishery. Fishery related factors may also impact on the seasonal nature of this fishery such as logistical problems that occur when operating from remote locations during the monsoon season (Nov-April). This includes access to fuel during the wet season. More recently native title claims excluding access to the inter-tidal zone have been lodged and granted for some areas.



**Figure 12.** Mean monthly catch in tonnes (closed histogram), effort traplifts/1000 (open histogram) and CPUE kg per traplift (open circle) for the Northern Territory Mud Crab Fishery 1999 to 2003.

## 5.2 Biological Influences on Catchability

Biological influences on catchability are also evident for this species. Monthly biological monitoring of the commercial catch demonstrates that sex ratios fluctuate throughout the year (Figure 13). Female mud crabs contribute 60 to 80% of the catch over the northern dry season (May-October), rapidly declining to around 10-20% at the onset of the monsoon. This cyclic pattern demonstrates a significant reduction in the catch of legal sized female mud crabs from the inshore regions of the Northern Territory fishery during the later stages of the monsoon season. This supports the work of Hill (1994) suggesting an offshore spawning migration where female mud crabs leave estuaries to seek suitable marine waters to spawn as increased monsoonal rainfall impacts on inshore salinities.

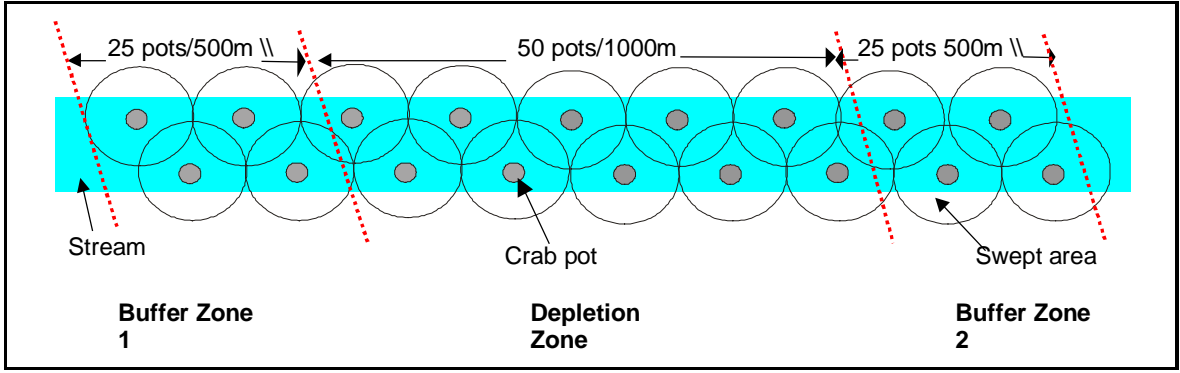


**Figure 13.** The percentage of female mud crabs in the Northern Territory commercial mud crab fishery catch August 2000-December 2004 (Northern Territory Department Primary Industry, Fisheries and Mines -Fishery commercial catch monitoring data).

Physical size also appears to influence the catchability of this species and additional investigation of subsequent behaviour of various size classes of crabs was required. To investigate the influence of large crabs on the capture of small crabs I designed an experiment that utilised mark-recapture and depletion techniques so comparisons could be made between the influence of replacement of tagged crabs and removal of crabs respectively.

### 5.2.1. Methods

In May 2000 a study was undertaken in a creek located adjacent to the mouth of the Adelaide River, approximately 60 kilometres east of Darwin (Figure 9). One hundred baited crab traps were spaced at 20m intervals, commencing at the creek mouth and alternating along each bank. Traps were individually identified using a numbered cattle ear-tag attached near the float and each trap site was marked using fluorescent flagging tape attached to the most adjacent mangrove tree along the creek bank. For eight days each trap was checked and rebaited every 24 hours with approximately 500 g fresh red meat (*Macropus* sp.) and re-set in the same location. The creek effectively was divided into three zones resulting in a one-kilometre depletion zone enclosed between two 500m mark-recapture (buffer) zones; buffer 1 was situated near the mouth of the stream (Figure 14).



**Figure 14.** Stylised diagram of depletion experiment including buffer (mark recapture) zones.

Traps were systematically checked every 24 hours on the daylight high tide and rebaited. Crabs caught within each buffer zone were sexed, assessed for moult stage; carapace measured, and tagged (Hill 1975; Robertson and Piper 1991) using individually numbered T-Bar tags (Hallprint Australia) before being returned to the water at the point of capture. Crabs caught within the depletion zone were measured, sexed, assessed for moult stage and removed from the study site for the duration of the experiment and returned live to the site on completion of the experiment.

As the same number of traps were set during each survey the numbers caught are directly proportional to the catch rate. To investigate the impact of size on catch, the catch was split into larger ( $\geq 120$  mm CL) and smaller ( $< 120$  mm CL) size classes in both the depletion and buffer zones. To standardise catches for both zones and size classes the catch for each size classes was initially standardised in each zone as a proportional change in catch compared to the first day of capture:

$$P_{1,i} = \frac{(n_i - n_1)}{n_1}.$$

Where  $P_{1,i}$  is the proportional change between the first and  $i$ th day,  $n_i$  is the number caught on day  $i$  and  $n_1$  is number caught on day 1.

To compared the relative change between size classes and zones the proportional change was further standardised by dividing the proportional change on subsequent days against the proportional change on the first day:

$$P_{std} = \frac{P_{1,i}}{P_{1,2}}$$

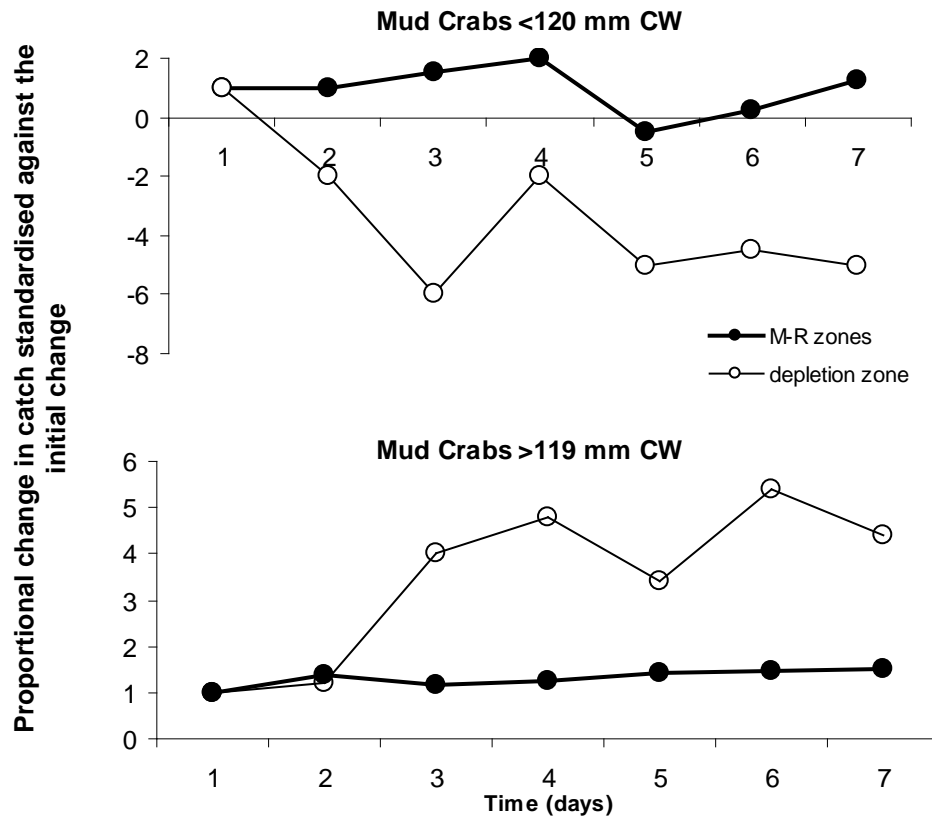
Note that a negative  $P_{std}$  value indicates a positive increase in catch between time periods.

### 5.2.2 Results

A total of 749 crabs were captured over the eight day period. 143 crabs were tagged and returned to the water in buffer zone 1 (500m downstream section of creek), 398 crabs were removed from the depletion zone (1000m) and 208 crabs were tagged and returned to the water in buffer zone 2 (500m upstream section of creek).

Catches from the buffer (mark-recapture) zone for both size classes of mud crabs showed only minor proportional changes throughout the eight days of the survey (Figure 15). In contrast,

larger crabs ( $\geq 119$  mm CW) declined steadily in the depletion zone as they were removed from this fishery. Smaller crabs ( $< 120$  mm CL) also showed a change by initially increasing during the first 3 days of the fishery and then remaining relatively constant at a proportionately higher catch.



**Figure 15.** Comparison of the daily proportional change in catch after standardisation against the initial change in catch for two size classes of mud crabs. There is limited change in the standardised catch from the buffer – mark-recapture zones (closed circles) whereas the catches from the depletion zone (open circles) shows substantial change during the survey period.

### 5.2.3 Discussion

This study provides evidence of an interaction between larger and smaller mud crabs. This is not surprising as *S. serrata* is known to be an aggressive predator. These results do have important ramifications for estimating abundance of smaller crabs from traps. Firstly, actual abundance estimates from trapping studies are unlikely to be accurate and will vary depending on the size composition of the population. Secondly, relative change in abundance of smaller crabs from trapping surveys will also be problematic as change in abundance can be due to both a change in actual abundance (e.g. recruitment) as well as catchability. This impact has been reported for lobsters by Frusher *et al.*, 2003. Further research is required to determine methods for standardising catch rates of smaller crabs against the catch rates or abundance of larger crabs.

For this study we split crabs into two size bins ( $< 120$  mm and  $\geq 120$  mm) and selected mark-recapture models that allowed for time dependent changes in each of the size classes.

## 5.3 Movement

An investigation of crab movement within this mark-recapture/depletion study design was required to evaluate the designs ability to meet the important assumption of a closed population, as movement (immigration or emigration) from the depletion zone would bias results.

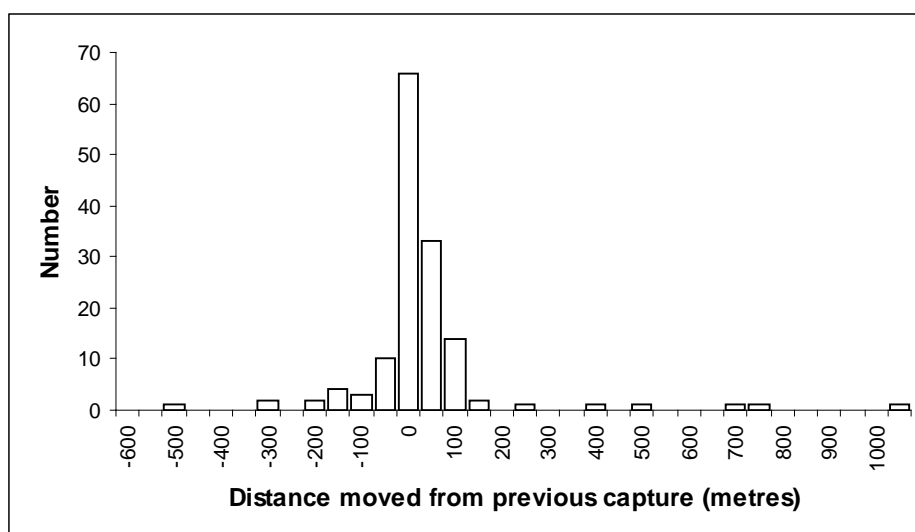
### 5.3.1 Methods

Using the same dataset collected in 2000 I was able to assess crab movement within the 2 kilometre section of stream under study. Each of the numbered crab traps (100) was positioned at 20 metre intervals along the creek bank and its position flagged with fluorescent tape in an adjacent mangrove tree. This enabled the movement of all tag recaptures to be plotted against the position of previous capture.

### 5.3.2 Results

351 crabs were tagged in the two buffer zones during the study and 132 individual tagged crabs were recaptured on 210 occasions. 82 tagged crabs were recaptured twice, 33 crabs were observed on three occasions, 7 crabs were recaptured four times, 8 crabs were recaptured on five occasions and one crab was caught on six and another seven times during the eight day study.

On preliminary review of the data a large proportion of recaptures (86.3%) were within 100m of previous capture location. To remove bias from the tag recovery analysis I disregarded capture and re-capture information from traps that did not have at least 100m of traps located adjacent to its position both upstream and downstream of the trap location (i.e. data from traps 1-5 and 95-100 were removed). Analysis of the revised data showed that 66 tagged crabs (46 % of total recaptures) did not move from their original capture location and were re-captured in the same trap (Figure 16). 88.1% of tagged crabs were recaptured within 100 metres of their previous capture location and 92.3% of recaptures were within 150m of the previous capture location. Movement was more predominant towards the mouth of the creek.



**Figure 16.** Distance individual tagged crabs moved from previous capture in metres (-ve values = upstream movement +ve values = downstream movement).



To investigate whether movement of tagged animals is influenced over time as the depletion zone is fished down I grouped the tagging data into day of capture and the recapture data into 4 day blocks recording first re-capture within the immediate 4 days after initial capture for the buffer zones and depletion zone (Table 4). In the buffer zones the proportion of crabs recaptured within 4 days of initial capture varied between 41.5 and 52.6. The difference was not significant ( $\chi^2$  1.6, df 3, p 0.659) indicating crab movement in the buffer zones is independent of day tagged. The proportion of crabs tagged in the buffer zones that moved into the depletion zone over the four day period varied from 2.2 to 5.3. This difference was also not significant ( $\chi^2$  0.765, df 3, p 0.858) indicating removal of tagged animals from the depletion zone does not increase movement of tagged buffer zone crabs into the depletion zone.

**Table 4.** Total number of individual crabs tagged during days 1-4 and first capture data for tagged crabs in buffer and depletion zones.

	<b>Number Tagged</b>	<b>Recapture period</b>	<b>Buffer Zones</b>		<b>Depletion Zone</b>	
			<b># recap</b>	<b>%</b>	<b># recap</b>	<b>%</b>
Day 1	75	Day 2-5	38	50.7	4	5.3
Day 2	46	Day 3-6	20	43.5	1	2.2
Day 3	38	Day 4-7	20	52.6	2	5.3
Day 4	41	Day 5-8	17	41.5	2	4.9

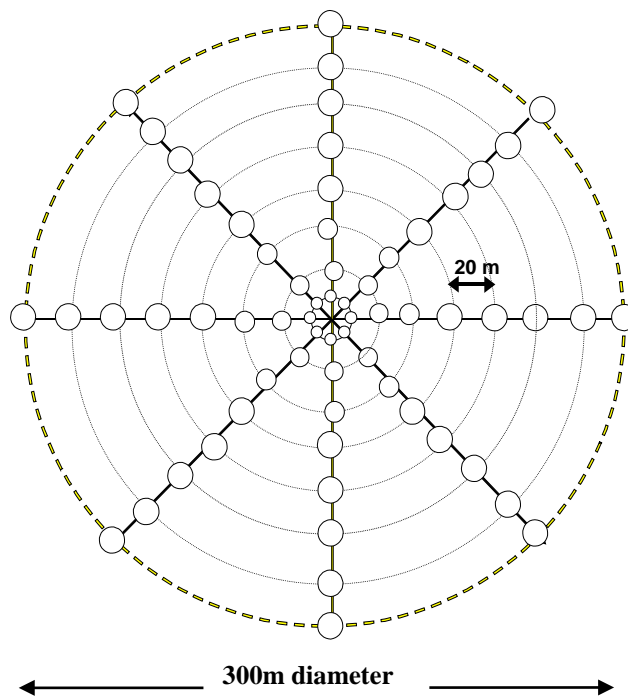
### 5.3.3 Discussion

During the 8 day study the majority of recaptured tagged animals remained within 100m of their previous capture. This is an important result demonstrating that this river based study design encompasses this species home range. The use of intensive trapping within river systems also does not appear to positively bias crab movement. The removal of animals from the central depletion zone does not encourage large scale immigration from the buffer zones supporting the assumption of a closed population during the eight day study period.

## 5.4 Preliminary Trapping Web Investigation

As trapping web theory had not been applied to the marine environment previously, various configurations of the web design needed to be tested to ensure assumptions of the analyses were met.

Using smaller variations on the design represented in Figure 17, four trapping web experiments were conducted on the foreshore areas of the Wearyan River in the southern GOC in 2001. Each experiment looked at different configurations of the trapping web design e.g. number of traps, spacing between traps and number of radial arms in the trapping web. A star picket marked the centre of the web and radial arms were set by compass bearing and each outer limit marked with an additional star picket and a bouy.



**Figure 17.** Stylised diagram of the trapping web design - 8 arms x 8 traps.

Trap spacing between pickets was set using a chainman measuring device, commonly used for measuring distances for geological applications. So as to concentrate traps at the web centre the inner ring of traps was placed at 10 m from the central point and the remaining traps were set at 20 m intervals along each arm of the web creating 8 concentric rings. Crab traps were individually identified by a numbered cattle ear-tag attached near the float. For five days each trap was checked and rebaited every 24 hours with around 500 grams of fresh red meat (*Macropus* sp.) and re-set in the same location. All crabs caught within the trapping web were sexed, assessed for moult stage; carapace width was measured, and tagged (Hill 1975; Robertson and Piper 1991) using individually numbered T-Bar tags (Hallprint, Australia) before being returned to the water at the point of capture.

### 5.4.1 Trapping Web Design

Due to the uncertainties in applying a new abundance estimation method for crustaceans, a number of trials were conducted to provide a design that best met the critical assumptions. As logistics permitted four small web designs were tested; designs were varied by trap number, trap spacing and the number of radial arms (Table 5).

**Table 5.** Summary of four trapping web designs tested in the southern GOC 2000-2001.

Web No.	Date	No traps	No arms	Trap Spacing (m)	Web Diameter (m)
1	11/10-15/10/2000	30	5	20	240
2	11/10-15/10/2000	30	5	40	480
3	15/09-19/09/2001	40	8	40	400
4	15/09-19/09/2001	40	8	60	600

#### 5.4.2 Trapping Web Analysis

The trapping web datasets were collected following Distance sampling theory (Buckland *et al.* 1993, 2001). Distance sampling is a widely-used group of closely related methods for estimating the density and/or abundance of biological populations. Distance sampling methods measure or estimate the distance an animal was detected from a line or point. This sampling method allows for the fact that some of the objects will go undetected and that there is a tendency for detectability to decrease with increasing distance from the observation point. Borchers *et al.* (2002), in reviewing methods for estimating animal abundance, identified one strength of Distance sampling theory as, “the ability to estimate abundance from a single survey due to the strong (and often reasonable) assumption about detection probability.”

Density estimates were produced using ‘distance sampling analysis’ (DISTANCE 3.5, Thomas *et al.* 1998). This specific software is dedicated to density estimation of free ranging animal populations. The semi-parametric modeling approach used in this software is based on the key + series adjustment method described by Buckland (1992). The key functions available in the software are the uniform, half-normal, hazard-rate, and negative exponential, and the adjustment terms are the cosine series, simple and hermite polynomials. The reader is referred to Buckland *et al.* (2001, p. 45-48) for further details about different models and series adjustments.

Data collected from each of the four trapping web layouts were analysed using the half normal + cosine adjustment and uniform + cosine adjustment models (Thomas *et al.* 1998). The model with the smallest Akaike's Information Criterion value was chosen (AIC, Akaike 1973; Thomas *et al.* 1998). AIC model selection uses a function minimisation framework and is based on the Kullback-Leibler "distance" between two distributions (Thomas *et al.* 2002; Thomas *et al.* 1998). Following the principle of parsimony the model that best fits the data with the fewest number of estimated parameters is selected. A delta AIC ( $\Delta AIC$ ) value of zero indicates the model of best fit and the corresponding estimated density of animals is given in number of animals per hectare.

As this design provides no physical barriers to emigration or immigration from in or outside of the study site (edge effect), it is generally suggested that the data be truncated to remove outliers and improve model fit.

#### 5.4.3 Results

For each analysis density estimates were calculated for the full dataset and also refined by truncating the outer one or two rings of each web to adjust for the edge effect (Buckland *et al.* 1993).

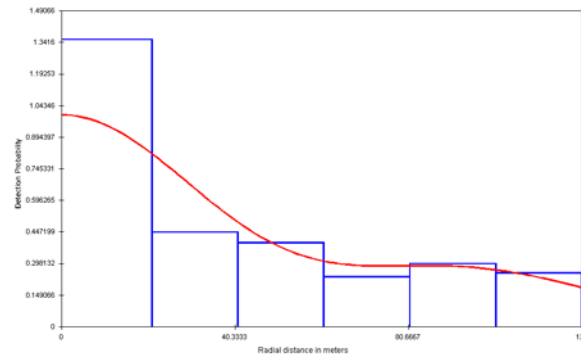
Web 1 applied 30 traps across 5 x 120m radial arms, with 6 traps spaced at 20m intervals along each radial arm. A total of 210 crabs were tagged and 130 tagged crabs were recaptured during the 5 day study. Plots of detection probability against radial trap distance for the full dataset and truncation of the outer one and two rings show improved fit with truncation (Figure 18).

Using first capture data a detection function was fitted to the datasets. The half-normal + cosine adjustment and uniform + cosine adjustment models have similar model fits and density estimate with density estimates ranging from 138 and 258 crabs per hectare depending on level of truncation (Table 6). The AIC model selection results indicate that the half-normal cosine model fitted the data best for both the full data set and truncate 2 data and the uniform cosine model was selected as best fit for the truncate 1 dataset.

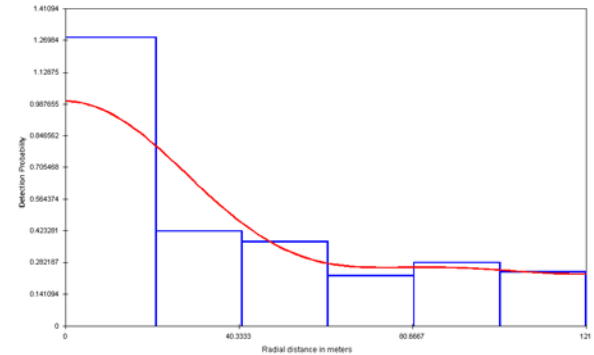
**Table 6.** Models, model selection statistics for trapping Web 1 using 30 traps set along 6 arms at 20m spacing, number of parameters (K)  $\Delta$  AIC, AIC, effective detection radius (EDR), e estimated density per hectare (D), detection probability (P) and coefficient of variation (CV) for first captures only.

Model	K	$\Delta$ AIC	AIC	EDR	D	D CV	P	P CV
Half-normal Cosine	2	0	749.52	69.48	138.48	0.192	0.32969	0.1786
Uniform Cosine	3	0.9	750.41	67.59	146.31	0.199	0.31205	0.1866
Half-normal Cosine trunc 1	2	6	521.08	56.50	160.51	0.216	0.31298	0.2011
Uniform Cosine truncate 1	4	0	521.02	49.25	211.28	0.279	0.23778	0.2677
Half-normal Cosine truncate 2	3	0	320.19	37.48	258.34	0.263	0.21409	0.2462
Uniform Cosine truncate 2	3	0.19	320.38	39.02	238.39	0.251	0.232	0.2328

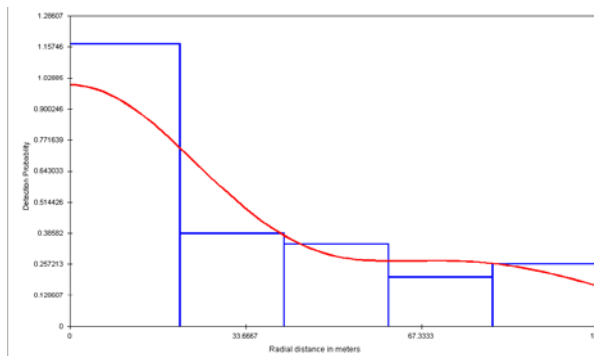
Half-normal Cosine Model



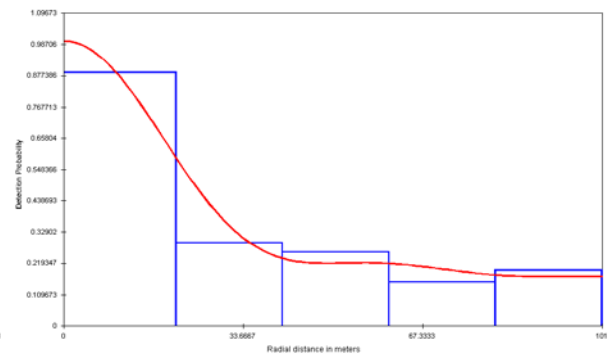
Uniform Cosine Model



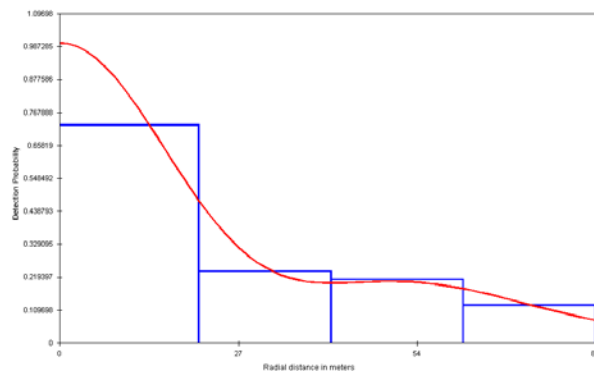
Half-normal Cosine Model truncate (1)



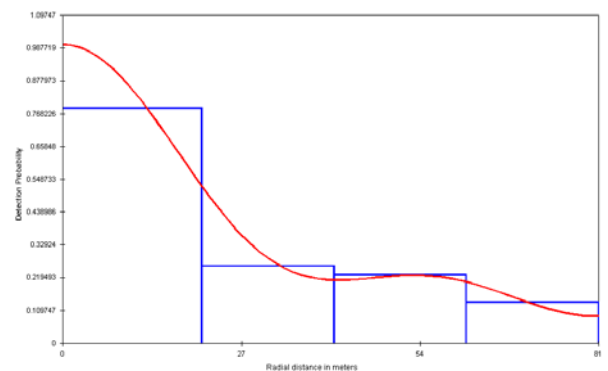
Uniform Cosine Model truncate (1)



Half-normal Cosine Model truncate (2)



Uniform Cosine Model truncate (2)

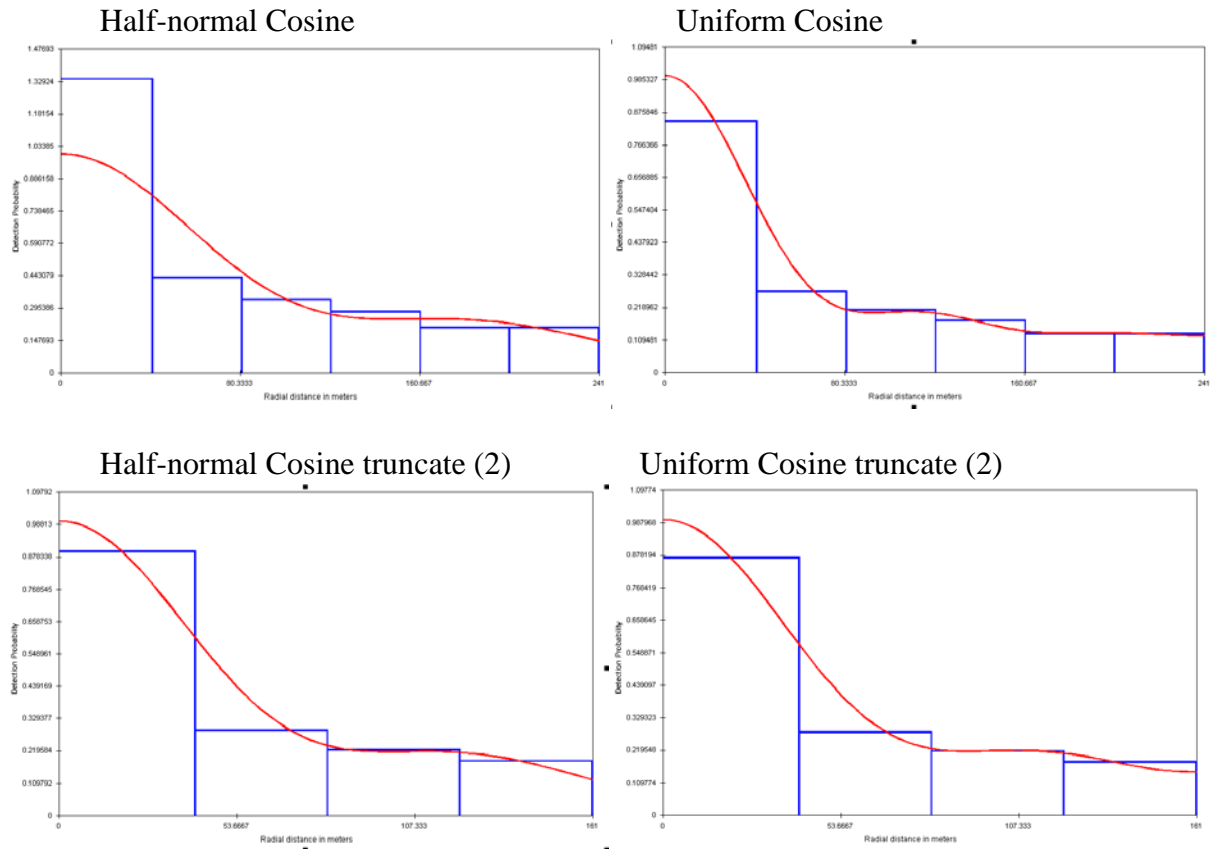


**Figure 18.** Detection probability for half normal and uniform cosine models for the full dataset and truncation of outer and outer two rings for web design 1 (30 traps, 6 radial arms, 20m spacing).

Web 2 applied 30 traps across 5 x 240m radial arms, with 6 traps spaced at 40m intervals along each radial arm. A total of 229 crabs were tagged and 128 tagged crabs were recaptured during the 5 day study. For each preferred model density estimates varied from 66 crabs/hectare to 84 crabs/hectare (Table 7). The AIC model selection indicated that half-normal cosine model fitted the data best for both the full data set and the truncated data. Detection probability plots for all analyses also provided (Figure 19).

**Table 7.** Models, model selection statistics for trapping Web 2 using 30 traps set along 5 arms at 40m spacing, number of parameters (K)  $\Delta$  AIC, AIC, effective detection radius (EDR), estimated density per hectare (D), detection probability (P) and coefficient of variation (CV) for first captures only.

Model	K	$\Delta$ AIC	AIC	EDR	D	D CV	P	P CV
Half-normal Cosine	2	3.4	827.37	130.09	43.07	0.173	0.28957	0.1584
Uniform Cosine	5	0	823.94	103.33	68.27	0.247	0.18224	0.2358
Half-normal Cosine truncate 2	2	0	388.32	82.74	64.63	0.226	0.26243	0.2079
Uniform Cosine truncate 2	3	1.2	389.56	81.50	66.61	0.245	0.25466	0.2286

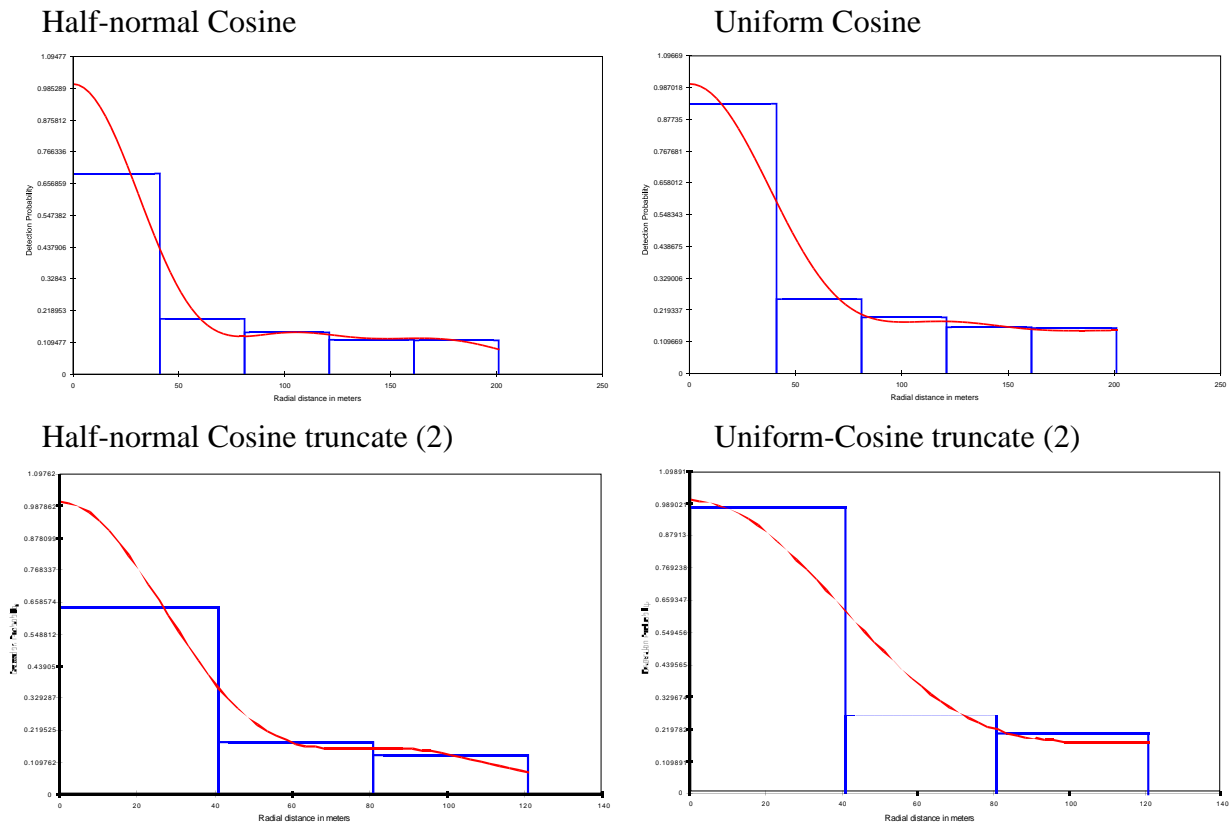


**Figure 19.** Detection probability for half normal and uniform cosine models for the full dataset and truncation of outer and outer two rings for web design 2 (30 traps, 6 radial arms, 40m spacing).

Web 3 applied 40 traps across 8 x 200m radial arms, with 5 traps spaced at 40m intervals along each radial arm. A total of 278 crabs were tagged and 52 tagged crabs were recaptured during the 5 day study. Density results for this web design varied from 78 crabs /hectare to 118 crab/hectare (Table 8). The AIC model selection indicated that half-normal cosine model fitted the data best for both the full data set and truncated data. Detection probability plots for the full dataset and truncated (2) datasets are also provided (Figure 20).

**Table 8.** Models, model selection statistics for trapping Web 3 using 40 traps set along 8 arms at 40m spacing, number of parameters (K)  $\Delta$  AIC, AIC, effective detection radius (EDR), estimated density per hectare (D), detection probability (P) and coefficient of variation (CV) for first captures only.

Name	K	$\Delta$ AIC	AIC	EDR	D	D CV	P	P CV
Half-normal Cosine	4	0	695.26	79.22	109.5	0.199	0.20926	0.2163
Uniform Cosine	4	3.8	699.07	91.94	81.32	0.227	0.15537	0.1873
Half-normal Cosine Truncate 2	2	0	249.38	54.93	118.1	0.215	0.31137	0.1757
Uniform Cosine Truncate 2	2	4.66	254.04	67.59	78.03	0.224	0.26647	0.2016

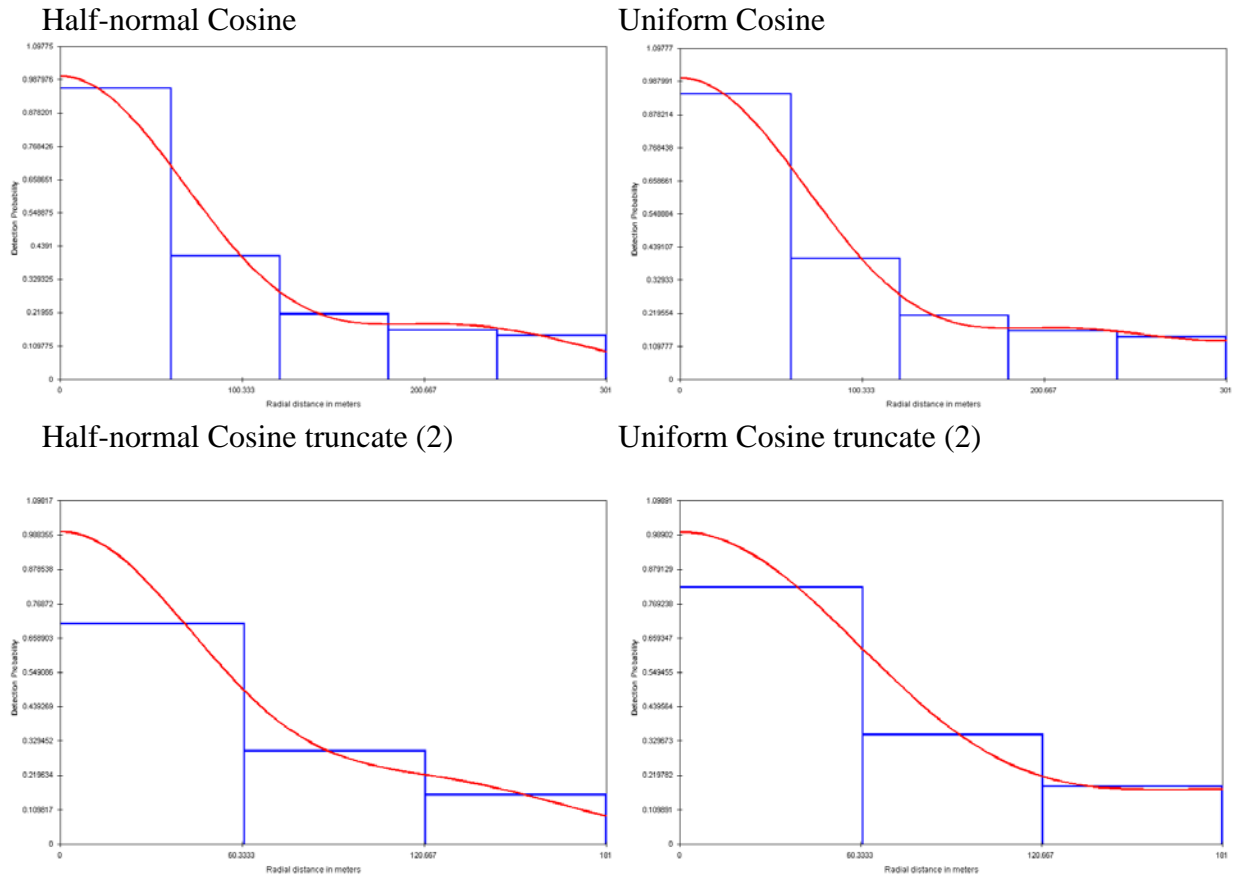


**Figure 20.** Detection probability for half normal and uniform cosine models including truncation effect for web design 3 (40 traps, 8 radial arms, 40m spacing).

Web 4 applied 40 traps across 8 radial arms with 5 traps spaced at 60m intervals along each radial arm. A total of 225 crabs were tagged and 80 tagged crabs were recaptured during the 5 day study. Density results for this web design varied from 40 crabs /hectare to 54 crabs/hectare Table 9). The AIC model selection indicated that half-normal cosine model fitted the data best for both the full data set and truncated data. Detection probability plots for the full dataset and truncated (2) datasets are also provided (Figure 21).

**Table 9.** Models, model selection statistics for trapping Web 4 using 40 traps set along 8 arms at 60m spacing, number of parameters (K)  $\Delta$  AIC, AIC, effective detection radius (EDR), estimated density per hectare (D), detection probability (P) and coefficient of variation (CV) for first captures only.

Name	K	$\Delta$ AIC	AIC	EDR	D	D CV	P	P CV
Half-normal Cosine	2	0	844.08	143.91	40.11	0.145	0.22859	0.1314
Uniform Cosine	2	1.34	845.42	142.92	40.67	0.146	0.22545	0.1331
Uniform Cosine Truncate 2	2	0.14	332.49	101	46.81	0.193	0.31137	0.1757
Half-normal Cosine Truncate 2	2	0	332.35	93.43	54.69	0.217	0.26647	0.2016



**Figure 21.** Detection probability for half normal and uniform cosine models including truncation effect for web design 4 (40 traps, 8 radial arms, 60m spacing).

#### 5.4.4 Discussion

Following the recommendations of Buckland *et al.* (2001) I varied web designs and analyses to review the impact of trap numbers, trap spacing and the level of truncation. Due to logistical restrictions the number of traps utilised in each web was considerably less than recommended, however I was mainly concerned in assessing overall trends.

The four web designs examined provided wide ranging density estimates (40 to 258 crabs/hectare).

Seven of the ten comparisons favoured the half-normal detection function with cosine smoothing indicating this was consistently the best performing model.

I compared the preferred model densities for the full dataset and truncated datasets for each web design. For Web 1 truncation by one interval (20m) increased the density estimate by 53% and truncation by two intervals (40m) increased the estimate by 87%. Truncation by two intervals, decreased the Web 2 estimate by 6%, increased the Web 3 estimate by 8% and for Web 4 the estimate varied by 36%. While Web 1 returned the highest density estimates this would suggest that Web 1 layout (240m diameter) with truncation was too small and failed to cover the home range of the study animal and this biased estimates positively. The diameter of Web 2 (480m) was double the diameter of Web 1 and this additional coverage appears to capture home range movement and not severely bias estimates. The Web 3 diameter of 400m also appears to perform well, while it appears trap spacing (60m) for Web 4 across the 600m web diameter introduces additional bias. Web 4 also produced the lowest density estimates.



Trapping web layouts with a web diameter >240m and >400m appear to provide the most reasonable estimates.

With Web 2 and Web 3 layouts providing the most reasonable estimates I next compared the coefficient of variation of the density estimates. Results suggest that Web 3 with smaller diameter (400m), increased number of traps (40) and 2 additional radial arms provided the greater precision.

The variability of the results observed in these studies is likely due to the small number of traps (30-40) and the subsequent expense of data truncation. Lucas *et al.* (1995) recommended a minimum use of 80-90 traps for a trapping web survey and on review the additional effort and logistical support needed to fish a larger number of traps is a likely requirement. For future studies I increased the total number of traps to 64, as this was the maximum number of traps we could logistically work with in the field and increase the number of concentric rings of traps from five to eight, effectively changing the web design to 8 radial arms each sampling with 8 traps.

## 6.0 ABUNDANCE ESTIMATION

The preliminary work thus described in this thesis evaluates the methods and important assumptions required for the estimation of crab abundance in two habitat areas, rivers and mud flats. In 2002 and 2003 three field trips per year were conducted during the dry season (May-October) at the Adelaide River and GOC study sites (Table 10).

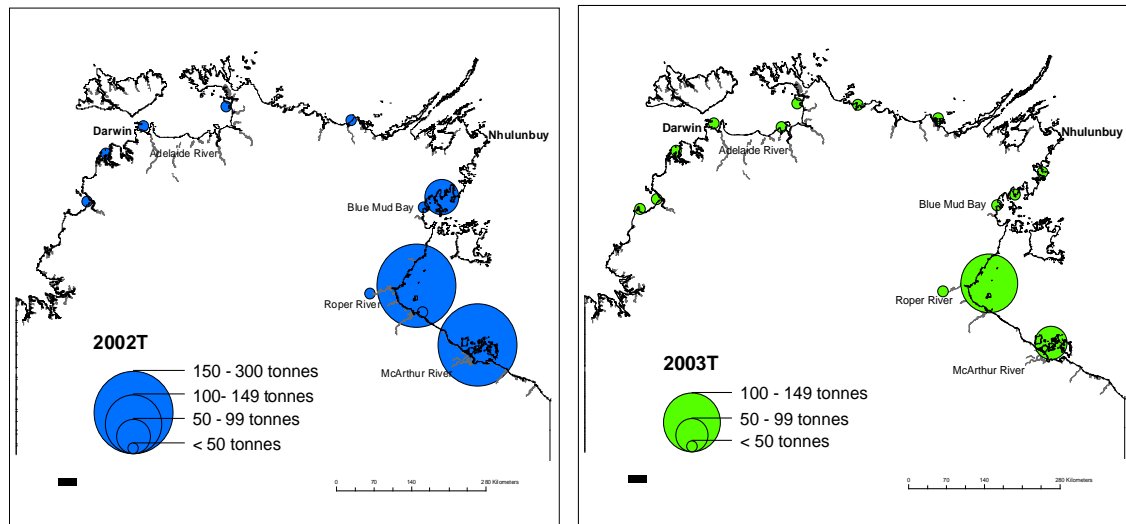
Each fieldtrip consisted of an eight-day depletion/ mark recapture experiment using 100 crab traps in a river system, and a five day trapping web study using 64 traps in an 8 line x 8 trap array on a foreshore mud flat. The two surveys were run simultaneously with the web usually scheduled to commence on day two of the mark-recapture/depletion study for logistical purposes.

**Table 10.** Summary of fieldwork conducted at the Adelaide River and the GOC in 2002 and 2003.

Adelaide River				GOC Wearyan River flats/Twin Sisters Creek			
Date	No Traps	Experiment	Completed	Date	No Traps	Experiment	Completed
3-10/5/02	100	Depletion	✓	20-27/05/02	100	Depletion	✓
5-9/5/02	64	Web	✓	22-26/05/02	64	Web	✓
1-8/07/02	100	Depletion	✓	16-23/06/02	100	Depletion	✓
2-6/07/02	64	Web	✓	17/06/02	64	Web	✗
30/7-6/08/02	100	Depletion	✓	25/8-1/09/02	100	Depletion	✓
31/7-4/08/02	64	Web	✓	25-29/08/02	64	Web	✓
				25-29/08/02	64	Web	✓
8/5-13/5/03	100	Depletion	✓	24-29/5/03	100	Depletion	✓
8-12/05/03	64	Web	✓	26-30/5/03	64	Web	✓
19-24/7/03	100	Depletion	✓	21-26/6/03	100	Depletion	✓
20-24/7/03	na	Web	✗	22-26/6/03	64	Web	✓
18-23/9/03	100	Depletion	✓	3-5/8/03	100	Depletion	✗
19-23/9/03	na	Web	✗	3-7/8/03	64	Web	✓

In 2002 all planned work was completed successfully. Unseasonal winds made access to the Wearyan River foreshore flats and the trapping web impossible in June 2002 and the experiment had to be rescheduled and was successfully completed in August 2002.

In 2003 a considerable drop in crab abundance at both study sites was evident, corresponding to a decline in reported commercial catch from both regions (Figure 3 and Figure 22). Trapping webs at the Adelaide River foreshore site were abandoned after May 2003 as capture and recapture rates declined to levels that made density estimation and the resources required to run each study unjustifiable. Extremely low crab numbers also resulted in the final Northern Territory GOC depletion study to be abandoned after day 3 when only three crabs had been removed from the 2 km study site.



**Figure 22.** Northern Territory commercial fishery mud crab harvest by region for 2002 and 2003.

Commercial crab traps are highly selective for legal sized mud crabs due to the large mesh size (75 x 25mm) used in their construction (Knuckey 1999). Research traps were constructed using a smaller mesh size (25 x 25 mm) to ensure capture of smaller size classes (Figure 23). During this study both fishing gear types were utilised at each study site. Due to tidal range differences and transport weight restrictions, different ratios of commercial to research traps were used at each site. At the Adelaide River site 25% of all gear used were commercial traps and at the remote Wearyan River (GOC) site 25% of fishing gear consisted of research traps.



**Figure 23.** Examples of commercial crab trap and research crab trap utilised at both study sites.

Little difference was observed in the proportion of small and large crabs caught between years for each gear type from the Adelaide River study (Table 11). The GOC data (where 75% of all traps were commercial- large mesh design) demonstrates the strong selectivity of commercial crab traps for large crabs and this changes little between years. The only contrast in this data can be found for the GOC research trap data. In 2002 35% of the total catch were crabs <120mm CW in comparison to 2003 when 74% of the total research trap catch were

crabs <120mm CW. This may be a function of reduced recruitment in 2002 and provides a likely explanation for the reduced commercial catch observed for the GOC in 2003.

**Table 11.** Proportion of small (<120 mm CW) and large  $\geq$  120 mm CW) crabs caught by gear type at the two Northern Territory study sites 2002 and 2003.

Gear Type	Crab carapace width (mm)	Adelaide R.		GOC	
		2002	2003	2002	2003
Commercial trap	% small crabs <120	17.65	14.88	7.65	9.66
	% large crabs $\geq$ 120	82.35	85.12	92.35	90.34
Research trap	% small crabs <120	59.09	54.10	35.54	74.44
	% large crabs $\geq$ 120	40.91	45.90	64.46	25.56

The two gear types described earlier were deployed in different proportions between study sites. During previous studies at the Adelaide River site, the large mesh and light-weight commercial crab traps were commonly lost, moved or the float submerged due to the large tidal range. To minimise the loss of gear and data, all Adelaide River studies used a 3:1 ratio of research (heavy-small mesh traps) to commercial traps. At the GOC study site, subject to a smaller tidal range, a reverse ratio was applied. Of the total traps 75% were of the commercial design and 25% of the alternate research gear type so that comparison of results could be achieved. Subsequently, differences in selectivity were evident with research traps being more selective for smaller crabs.

## 6.1 River Estimates

### 6.1.1 Depletion/Mark Recapture Survey Methods

Using the same mark-recapture/depletion study design described earlier in Chapter 5 (5.2.1) one hundred baited crab traps were spaced at 20-m intervals, commencing at the creek mouth and alternating along each bank. Traps were individually identified using a numbered cattle ear-tag attached near the float and each trap site was marked using fluorescent flagging tape attached to the most adjacent mangrove tree along the creek bank. For eight days each trap was checked and rebaited every 24 hours with approximately 500 g fresh red meat (*Macropus* sp.) and re-set in the same location. The creek effectively was divided into three zones resulting in a one-kilometre depletion zone enclosed between two 500-m mark-recapture (buffer) zones; buffer 1 situated near the mouth of the stream, although there were no physical barriers between zones (Figure 14). The Adelaide River depletion study site had an average width of around 30 m while the GOC depletion river site was around 70 m wide.

### 6.1.2 Depletion/Mark Recapture Data Analysis

Data collected in this study were analysed using the Huggins closed capture model (Huggins 1989, 1991). This model permits the estimation of population size when capture probabilities are heterogeneous, by modelling the capture probability in terms of observable covariates, in this case size. The approach used in the Huggins' model is equivalent to the Horvitz-Thompson sampling design, where animals have unequal probability of being included in the sample. To enable detection of differences in size related behavioural patterns, all data was

formatted into two groups, small immature animals (<120 mm CW) and large animals  $\geq$  120 mm CW); those that were mature or would reach maturity within the next moult. Each individual carapace width was also applied as a covariate using the Huggins closed capture model.

The model selection process used in this study is based on parsimony is outlined by Burnham and Anderson (1998). Parsimony is assessed using Akaike's information criterion (AIC). Where two models return similar results, the model with the smallest delta AIC value is selected. This process offers a compromise between goodness of fit and the number of parameters (Borchers *et al.* 2002). All models were fitted using the program MARK (White, 2003)

The results support the hypothesis of size influencing capture probability. The analysis consistently resulted in the selection of two models describing different capture probabilities for small and large crabs caught within the mark-recapture (buffer) and the depletion zones.

For both study sites and years the capture probability ( $p$ ) for crabs caught within buffer zones (where tagged animals were returned to the water at point of capture) was consistently best described by the models  $p(.)$  corresponding to the capture probability of small and large crabs remaining constant throughout each study. Recapture probabilities ( $c$ ) for buffer zones resulted in either model  $c(.)$ ,  $c(g)$  or  $c(t)$ . The first model  $c(.)$  corresponds to constant probability of recapture across both time and group indicating no influence on recapture. Selection of model  $c(g)$  indicated that probability of recapture was influenced by group (in this case large crabs) or  $c(t)$  where the probability of recapture changes with time.

For analyses of depletion zone data where there were no recaptures, probability of recapture ( $c$ ) was set to zero (i.e.  $c=0$ ). The most parsimonious full models in this case were,  $p(\text{size})$   $c=0$  and  $p((\text{small}^*t, \text{large}(.))$   $c=0$ , indicating that the probability of capture is primarily influenced by crabs larger than the mean size or the probability of capture of small crabs increases over time (as large crabs are removed), while remaining constant for large crabs.

Density estimates for small (< 120 mm carapace width) and large ( $\geq$  120 mm carapace width) crabs from buffer and depletion zones are presented for the Adelaide and Wearyan sites in Appendix 2 (Table 15). Densities were standardised as number of crabs per kilometre of stream and converted to number of crabs per square kilometre. Average density estimates for all crabs and for large crabs, for each year were calculated as the sum of estimates for the buffer zones plus depletion zone for each survey in each year at each site.

### **6.1.3 River Abundance Estimation Results**

A total of 2040 crabs were captured during the 2002 and 2003 sampling periods and a total of 700 recaptures were recorded during the study period.

A closed population is a key assumption of the mark-recapture and depletion data analyses undertaken in this study. For this to be true, movement between the buffer and depletion zones needs to be minimal. This study design allows measurement of the total number of crabs tagged in each buffer zone and movement (immigration) of any tagged animals from each buffer zone into the depletion zone. In Chapter 5.3 I demonstrated very limited movement occurred between the capture zones during the Adelaide River May 2000 survey. To examine for any seasonal movement patterns I examined the movement patterns of tagged crabs from 11 surveys conducted in 2002 and 2003. These results support my preliminary

findings and previous studies that mud crabs show limited movement (Table 12). Despite these low values there were indications of consistent trends between years in the extent of movement in the Adelaide River. Movement was less for all 3 survey times in 2003 and the greatest movement occurred in May during both years. In the GOC the small sample size in 2003 prevented comparisons between years. However, in 2002 the July sample was undertaken in June and produced a substantially higher movement rate. Examination of the biological data indicates that that all 17 crabs were intermoult males, greater than 120 mm carapace width. A pattern of movement involving one sex, all of similar physiological condition, may indicate the onset of a biological function such as moulting. These crabs were removed from the study site and stored alive at camp. To avoid over estimation of crab abundance any tagged animals that moved into the depletion zone were not included in the depletion zone catches during analyses.

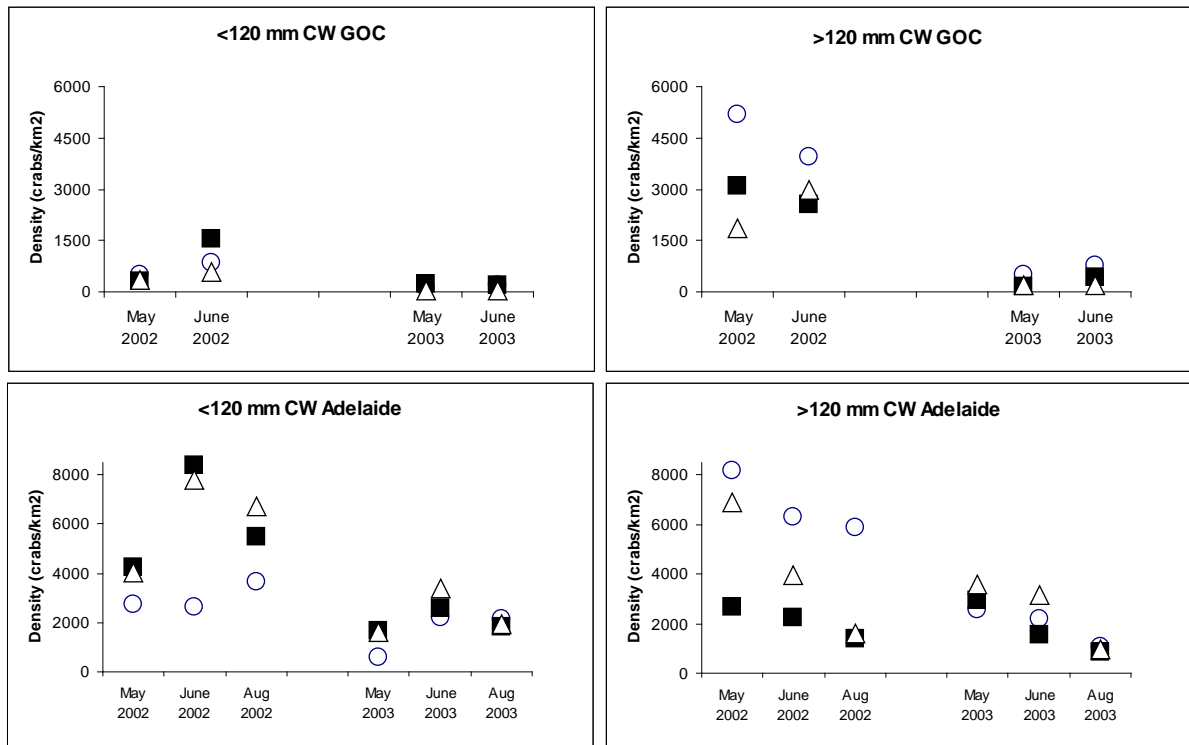
**Table 12.** Number of crabs tagged during each study in buffer zone 1 and buffer zone 2 and the number and percentage that moved into the depletion zone.

Site	Year	Month	Total number crabs tagged		Number and % tags moved	
			Buffer1	Buffer 2	Depletion	% movement
Adelaide River 1	2002	May	97	130	8	3.52
Adelaide River 2	2002	July	93	114	4	1.93
Adelaide River 3	2002	Aug	84	89	4	2.31
Adelaide River 4	2003	May	40	63	2	1.94
Adelaide River 5	2003	July	58	75	0	0.00
Adelaide River 6	2003	sep	41	33	1	1.35
GOC 1	2002	May	100	44	4	2.78
GOC 2	2002	June	110	74	17	9.24
GOC 3	2002	Aug	37	57	2	2.13
GOC 4	2003	May	15	9	0	0.00
GOC 5	2003	June	32	7	1	2.56

A general trend of reduction in crab numbers is evident between years for each site, especially at the GOC river study site (Figure 24).

**Figure 24.** Length-frequency distribution of male and female mud crabs from Adelaide River and GOC mark-recapture and depletion studies 2002-2003.

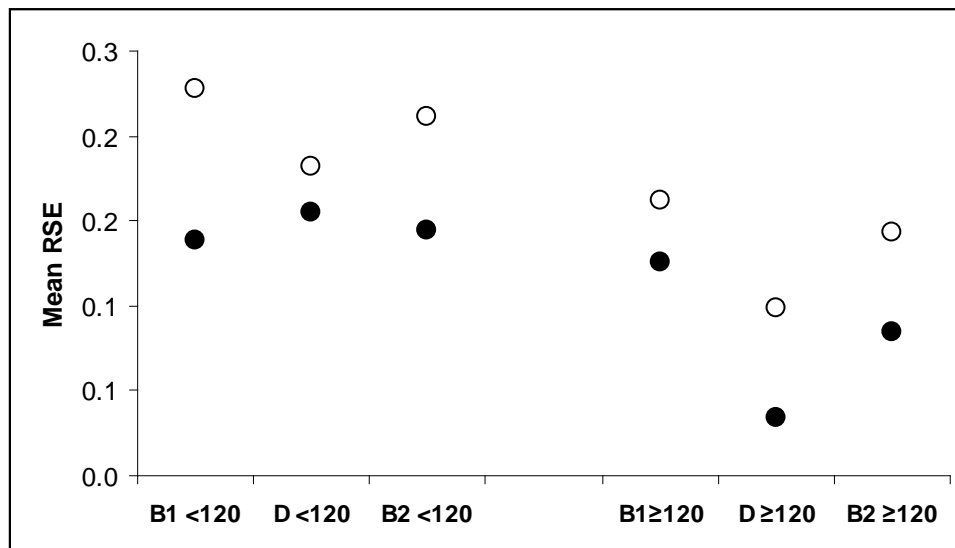
The Adelaide River density estimates for the river studies are lower than the GOC river estimates for both size groups (Figure 25). There is a consistent trend of reduced density from 2002 to 2003 for both size groups, both regions and each area/method except for the crabs  $\geq 120$  mm CW from the Adelaide River depletion zone. Large crabs (RHS panels) generally show a decline in density from May-Jun-Sep and this is expected as they are being commercially fished over the year. Conversely, small crabs show an increase from May to June/July.



**Figure 25.** Comparison of river density estimates from each capture zone (Buffer 1 =open circle, depletion closed square, buffer 2 open triangle) for each size category for the GOC and Adelaide River study sites in 2002 and 2003.

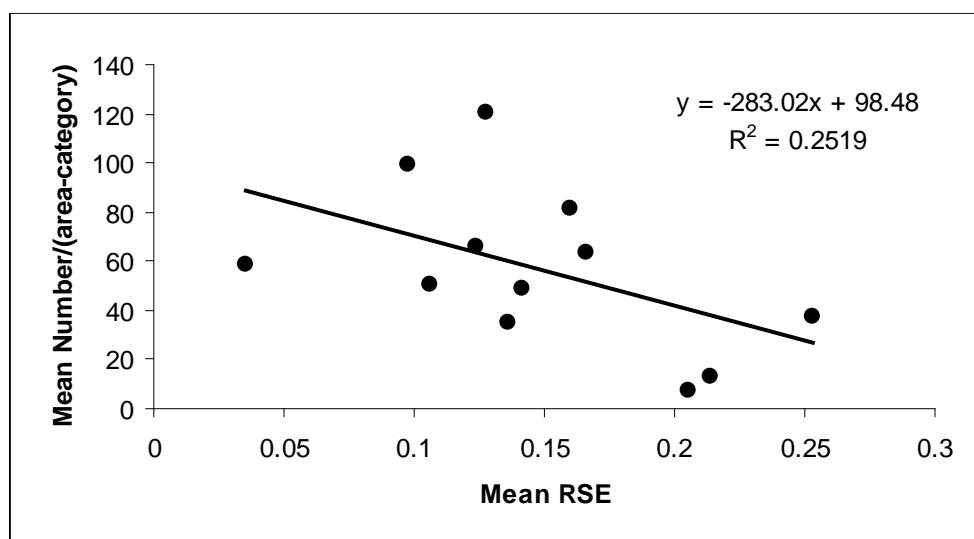
In testing the precision of the data analysis I also plotted the RSE for the different categories (capture zone and size class) by region (Figure 26). The RSE are consistently lower for the Adelaide River and the RSE of larger crabs is consistently lower for each area/method in both regions.





**Figure 26.** Mean RSE of capture zone (B1=Buffer 1, D=depletion, B2=buffer 2) for the Adelaide River (closed circle) and GOC (open circle) and size class (<120mm CW and ≥ 120mm CW).

I also was interested in the effect of sample size on estimate precision. I plotted the mean RSE against the mean catch for each capture zone and size class across both regions. This resulted in a negative linear regression with a correlation coefficient of -0.5. This suggests that sample size has some impact on the estimates. It is likely that the greater selectivity of the research traps for small animals at the Adelaide River site is responsible for the lower RSE values calculated for this region (Figure 27).



**Figure 27.** Influence of sample size for each capture zone and size class on estimate precision.

#### 6.1.4 Discussion

When the density estimates from the two study sites are compared it is obvious that the two systems are performing very differently. Crab density in the Adelaide river system is generally greater than that found in the Gulf river system. Caution needs to be applied in evaluation of differences in small crab abundance (<120mm CW) between sites due to the differences in fishing gear selectivity for small animals. As mentioned previously the Adelaide River region is subject to large tidal movement whilst the GOC is not. If the tidal

range does inhibit the habitat area that crabs prefer then the foreshore flat estimates should also demonstrate a similar pattern.

There is a consistent trend of reduced density from 2002 to 2003 for both size groups, for both regions and each area/capture zone except for May 2003 Adelaide River depletion zone (D)  $\geq 120\text{mm}$ . The decline in catch between 2002 and 2003 is also supported by the commercial fishery data. Large crabs generally show a decline in density from May-June-September and this is likely an effect of fishing pressure on legal size animals. Conversely, small crabs show an increase from May to June/July suggesting either an increase in catchability or recruitment to the fishery.

## **6.2 Foreshore Mud Flat Estimates**

### **6.2.1 Methods**

Following the results of the preliminary studies sixty-four baited crab traps were set in a trapping web design on the foreshore mud flats at the two study sites previously described. Each trapping web consisted of eight arms of eight crab traps all radiating from a star picket that marked the central point of the web. The web consisted of eight concentric rings of crab traps (Figure 17). Trap spacing is uniform for rings 2-8 with the inner ring spaced at  $\frac{1}{2}$  of this distance to ensure maximum trap saturation within the central web zone. Each web diameter was 300m and sampled an area of 70650 square metres. Star pickets and coloured buoys marked the end of each radial arm.

For five consecutive days, traps were systematically checked on the daylight high tide and rebaited. Crabs caught were sexed, assessed for moult stage; carapace measured, and tagged (Hill 1975; Robertson and Piper 1991) using individually numbered T-Bar tags (Hallprint Australia) before being returned to the water at the point of capture.

### **6.2.2 Data Analysis**

Two methods of analyses of trapping web datasets were compared. Both analysis methods allowed the use similar detection models. The half normal and uniform models were selected for both analyses.

#### **6.2.2.1. Distance Analyses**

Analysis of trapping web data using Distance sampling theory was described in Chapter 5.4.2.

Data were analysed using DISTANCE sampling software, Version 3.5 (Thomas *et al.* 1998.) Two standard Distance sampling models were chosen for the trapping web analysis; the half-normal with cosine adjustment model and the uniform with cosine adjustment models (see Buckland *et al.* 1993). The probability of detection was determined as the ratio of recapture to new capture of crabs in the innermost ring on each day of the survey, fitted using a linear optimiser routine (Excel SOLVER). To account for the effects of immigration from outside the study site (the edge effect) the data was truncated, with data from captures recorded from the outer two rings removed from the analyses.

Due to the small sample sizes observed for a number of the 2003 trapping web studies, numbers of crabs caught in the inner two rings were pooled and the initial interval distance from the centre of the web was subsequently adjusted from 10 m to 30 m for these analyses.

#### **6.2.2.2 Density Analyses**

Efford (2004) proposed a theoretical model of the trapping process that avoids edge effect adjustment and effective trapping area calculations. This new method views captures within a known trap design as a function of density and a 2-parameter capture function. These three parameters can be estimated at the same time using inverse prediction and simulation (Brown, 1982; Carothers, 1979; Pledger and Efford 1998; Efford *et al.* 2004), using general capture-recapture statistics as predictors (Efford *et al.*, 2004).

In brief the estimated population density ( $D$ ) is calculated using inverse prediction and Monte Carlo simulation. The parameter  $D$  describes the intensity of animal range centres as a spatial point process. Similar to Distance analyses, a spatial detection function  $g(r)$  describes the probability of capture as a decreasing function of distance between the detector (trap) and the animal's home range centre. To enable these calculations two additional parameters of an individual's capture probability ( $g$ ) are required. These correspond to measures of home range size ( $\sigma$ ) and susceptibility to capture ( $g_0$ ). The method also utilises information gained from conventional closed population estimates, population size ( $N$ ), mean capture probability ( $p$ ) and also incorporates the spatial information of the mean distance between successive recaptures of an individual ( $d$ ).

Efford's method is capable of analysing data from any configuration of traps (e.g. grid, web or line) and provides the option to choose from the standard closed-population estimators. I chose the jackknife estimator (Burnham and Overton 1978) as the author reports this estimator appears robust to heterogeneity and free from bias.

The assumptions for estimating density by inverse prediction are:

1. The population is closed (i.e. there are no births, deaths or dispersal events during a trapping session - of particular relevance to this species is the absence of recruitment to the site including both movement and moulting).
2. Capture does not affect the pattern of movement of an animal within a trapping session.
3. Tags are not lost, and the recapture data is recorded accurately.
4. Traps are set at known locations for a fixed time.
5. Trap placement is random with respect to the location of animal ranges.
6. Animals occupy home ranges that do not change during a trapping session.
7. Home ranges are roughly circular.
8. Home range centres are scattered throughout the area sampled, or home range centres are scattered within a mapped subset of the landscape.
9. The chosen closed population estimator is robust to other variation in capture probability (e.g. temporal variation).

Analyses were performed using the Windows® program DENSITY (Efford 2003). A graphic interface permits visualisation of spatial movement of individual tagged animals within the study site by individual animal and day of capture. This analysis utilises both capture and recapture data. The software was unable to fit models to the data from Wearyan August 2002, due to extremely low recapture rates (4 recaptures from 119 tagged animals).

DENSITY analyses legend (see Table 16 and Table 17 Appendix 2)

$\hat{N}$	Population estimate
Se $\hat{N}$	Estimated standard error ( $\hat{N}$ )
$\hat{p}$	Daily capture probability implied by N-hat, given the data
$\bar{d}$	Mean distance between successive captures (m) pooled over individuals
Se $\bar{d}$	Standard error ( $\bar{d}$ )
$D$	Density by inverse prediction (see above for units)
Se $D$	Standard error (Density) (prediction Se)
$g_0$	Core trapability estimated by inverse prediction
$\sigma$	Spatial scale of detection (m) by inverse prediction

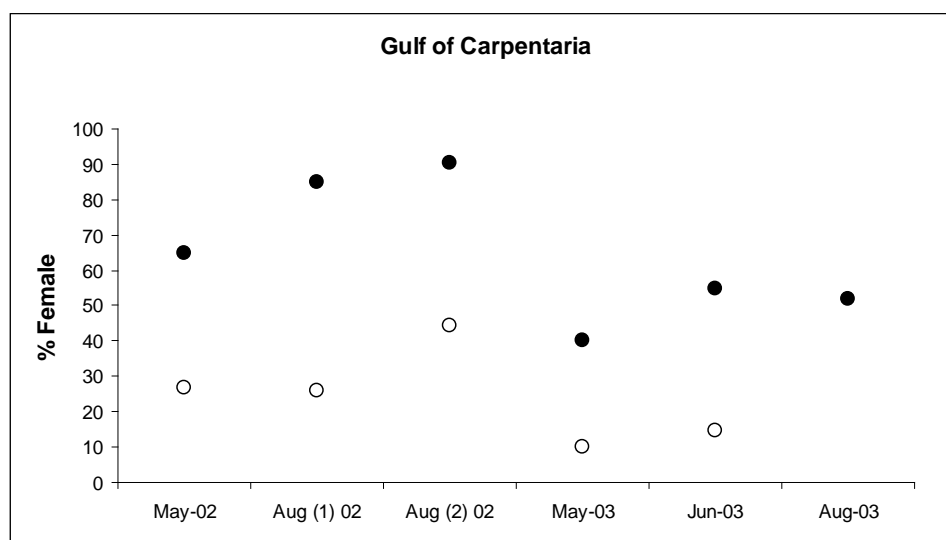
For each study both half-normal and uniform distributions were trialled.

### **6.2.3 RESULTS**

A total of 1169 crabs were captured during the 2002 and 2003 sampling periods and a total of 176 recaptures were recorded during the study period. Differences in the numbers of crabs caught from the foreshore flats at the two study site were evident with relatively few crabs caught on the Adelaide River foreshore flats (Figure 28).

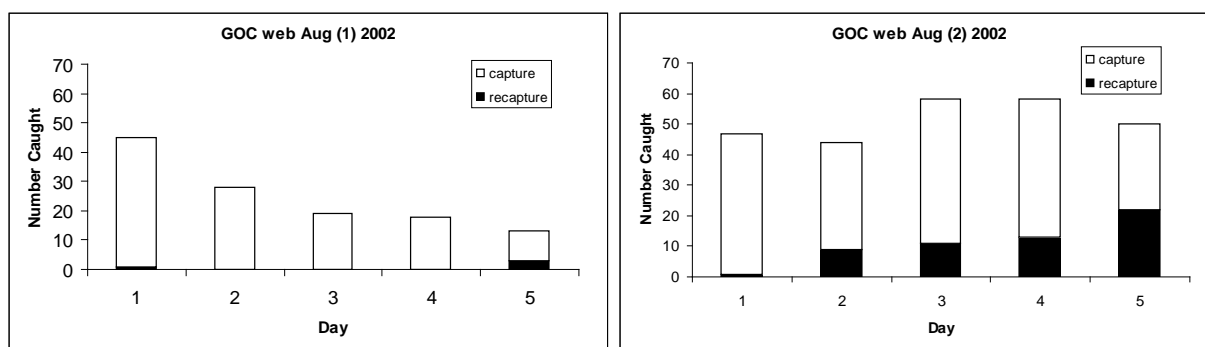
Differences in sex ratios were observed in the GOC trapping studies. The percentage of female crabs captured on the mud flats was considerably higher than that observed in the rivers during all sampling occasions. Similar patterns were observed in the magnitude of percent females in the catch between the river and foreshore mud flat areas. This pattern was not observed for the Adelaide foreshore flats dataset (Figure 29).

**Figure 28.** Length-frequency distribution of male and female mud crabs from Adelaide River and GOC trapping web studies 2002-2003



**Figure 29.** Comparison of the sex ratio (% females) for rivers (open circle) and foreshore mud flats (closed circle) in the GOC during surveys conducted in 2002 and 2003.

The Adelaide River foreshore flats trapping web studies were not continued after May 2003 due to extremely low catch rates and the costs associated with continuing the work could not be justified to the state fisheries department. In June 2002 unseasonal winds kept the tide out and traps in the trapping web were inaccessible for two days resulting in the abandonment of this survey. I rescheduled the trapping web and carried out two webs simultaneously in August 2002. Unfortunately this stretched resources and one of the webs was placed too close to the mangroves and traps were left high and dry on the low tide, making the traps only available to crabs on the high tide. Very different capture and recapture patterns were observed (Figure 30). While this may simply indicate variability in the method it is likely the web underperformed due to poor site selection and excess crab movement.

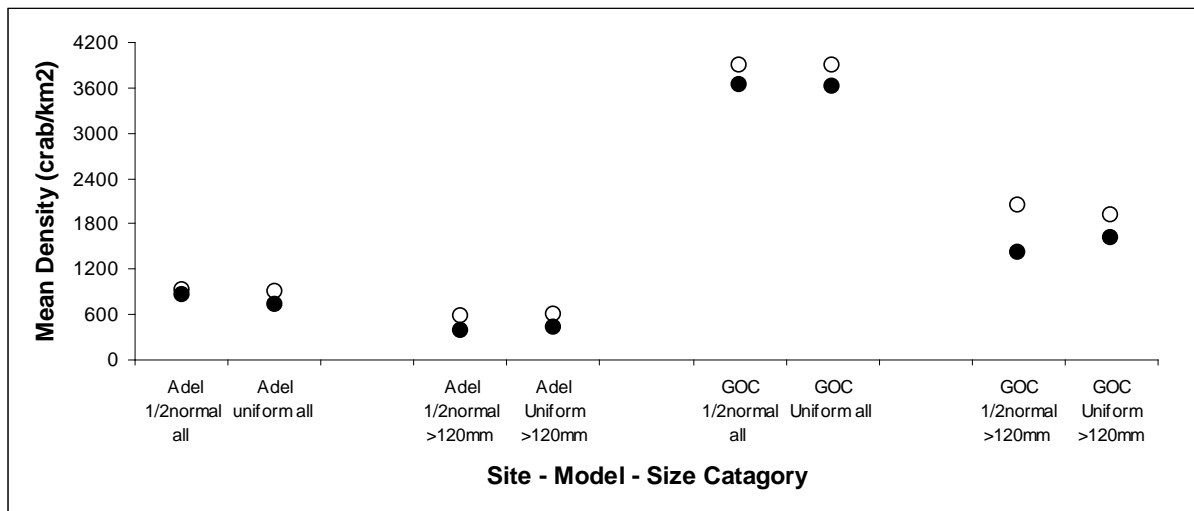


**Figure 30.** Impact of web placement on foreshore flats on capture and recapture rates.

Estimates of mud crab density obtained from the foreshore flat areas using trapping web surveys are provided in Appendix 2 (Table 16) and estimates of crab density from foreshore areas for large crabs  $\geq 120$  mm are provided in Appendix 2 (Table 17). These tables provide a comparison of results from the two analysis methods used in this study, DISTANCE and DENSITY analyses. A major data difference between the two analysis methods is that DISTANCE uses only first capture information while DENSITY utilised both capture and recapture information. Both the uniform and half normal detection function models were applied for each of the analysis methods. It was possible to resolve all the analyses using both detection function models using DISTANCE. The uniform detection function was required to

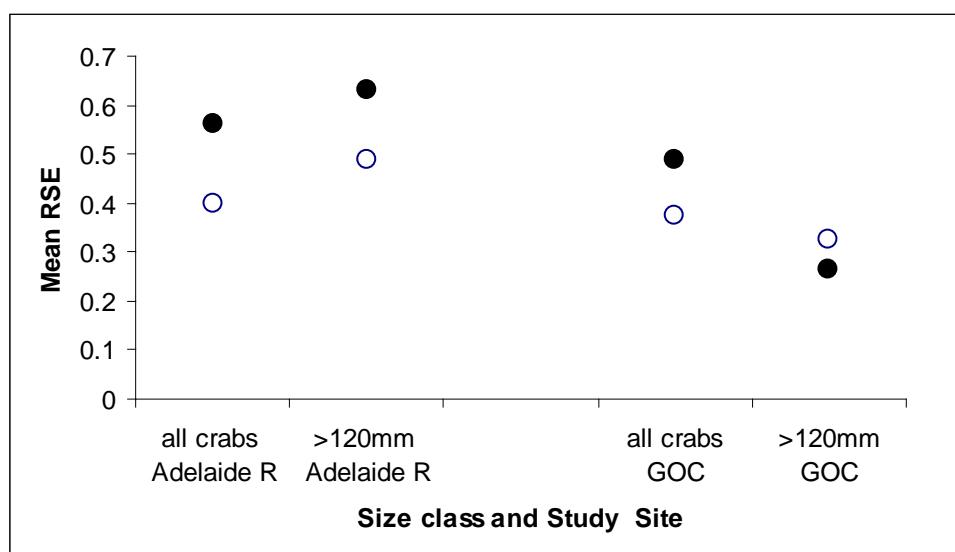
fit data with DENSITY with one dataset unable to be analysed due to the sparse data set previously discussed (GOC 1 August 2002).

I plotted the mean density estimate obtained from each detection model (half normal and cosine) for both analysis methods (Figure 31). The half normal and cosine detection models performed equally well for both Distance and Density analyses. The density of crabs found on mud flats differs greatly at both sites. The Adelaide River mud flats hold substantially less crabs than the GOC site. DISTANCE analysis produces a slightly higher density estimate than DENSITY analysis. The Adelaide mud flats hold relatively few small crabs whereas the GOC site demonstrates similar proportions of large and small crabs.



**Figure 31.** Comparison of mean density of all crabs (crab/km<sup>2</sup>) and crabs  $\geq 120$ mm CW using two detection models for DISTANCE (open circle) and DENSITY (closed circle) analysis.

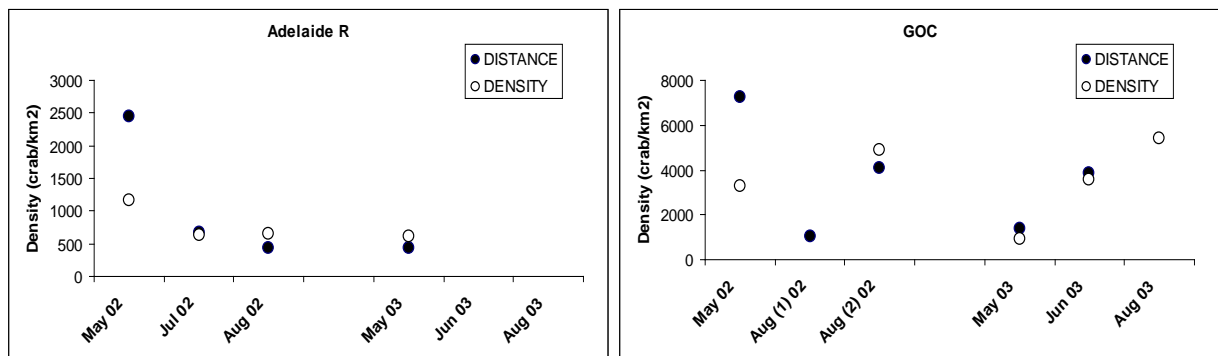
I then examined the precision of the estimates obtained from the two analysis methods by plotting the mean relative standard error (Figure 32). In 3 out of 4 occasions DISTANCE estimates returned a lower RSE than DENSITY.



**Figure 32.** Comparison of the mean relative standard error of DISTANCE (open circle) and DENSITY (closed circle) estimates for all crabs and large crabs ( $\geq 120$ mm CW) from the Adelaide River and GOC.



I compared the density estimates obtained by both analysis methods for each trapping web study by site (Figure 33). The decline in density observed between years for the river datasets is not apparent for the foreshore mud flat estimates. The GOC density estimates from 2002 show no clear pattern, even when the result from Aug (1) 2002 is discounted, while the 2003 estimates show a consistent increase in density over the sampling periods.



**Figure 33.** Comparison of density estimates (crabs/km<sup>2</sup>) by sampling period, site and analysis method.

#### 6.2.4 Discussion

The densities of crabs on the GOC foreshore mudflats are substantially higher than the Adelaide River mudflats, and this is reversed for the river systems with higher densities found in the Adelaide River system than the GOC River.

The difference in the results from the two trapping webs conducted concurrently in August 2002 provided a good example of the need to understand and meet the analysis assumptions. One web was located in an area that did not remain submerged for the full tidal cycle causing excessive movement of crabs as the tide receded and this resulted in reduced capture and almost nil recaptures.

The results of this work indicate that mud crabs are not uniformly distributed throughout the coastal environment. Males are more common in the river regions and females on the mudflats. The drivers of this behaviour are unknown and while both sexes commonly demonstrate aggressive behaviour, adult males generally grow larger, exhibiting larger chelae. This spatial division of the sexes may increase the probability of female survival.

There was a substantial decline in the Northern Territory commercial catch between 2002 and 2003. This was reflected by a decline in the river crab density estimates but was less evident in the density estimates from the mud flat data. For both the Adelaide River and the GOC, the relative change in catch rates appears to be better supported by the models applied in the DISTANCE estimates than the estimates obtained through simulation and inverse prediction (DENSITY estimates). Both the catch rates and the density estimates using the models applied in the DISTANCE analysis support the observed decline in commercial catch rates. In the GOC there was a substantial decline in the proportion of female crabs in the catch between 2002 and 2003 in both the river and mudflat regions suggesting that a lack of legal sized females was a major cause of the decline in catch. Although environmental variables are considered to be major drivers of recruitment in the GOC, the decline in abundance together with the decline in the proportion of female mud crabs is of concern for future recruitment to the fishery in 12 to 18 months time.

The trapping web design used in this study evolved from the preliminary studies in 2000/01 when commercial catch rates and crab density were at a fishery high. It is possible that as

density decreases the web design and trap spacing need adjustment due to the effect of reduced competition for traps.

## 6.3 Abundance Estimates for Study Sites

### 6.3.1 Introduction

For the abundance estimates to be of value for management of the Northern Territory mud crab fishery it is necessary to determine the abundance for the same spatial scale and measurement unit (tonnes) that the catch is recorded. In the Northern Territory catch is reported by fishers in 60 nm x 60 nm fishing (Appendix 1).

### 6.3.2 Methods

Abundance estimates for the rivers were based on mark-recapture depletion estimates using MARK software (Chapter 6.1.2) and for the mud flats; estimates were obtained from the trapping web using DISTANCE software (Chapter 6.2.2.1). These estimates were then extrapolated to the spatial scale of the fishing block by determining the area of mangrove lined streams and mudflats in each block (Chapter 3.3). Estimates of crabs greater than 120 mm CL were considered to represent the commercial catch as they were either legal sized or expected to become legal sized within the next moult (which would occur during the fishing season).

The abundance estimates were recorded as number of crabs and were converted to tonnes for comparison with the commercial catch. The Department of Primary Industry, Fisheries and Mines monitors the commercial catch from four regions in the Northern Territory for biological changes in catch composition. Each month a series of biological measurements from a random sample of 100 crabs from four regions are collected. The mean weight and size of mud crabs from the Adelaide River and GOC regions have declined between 2002 and 2003 indicating that either the fishery captured a greater number of recruits or fewer larger crabs were available for capture in 2003 (Table 13).

**Table 13.** Mean weight (g) and mean carapace width (CW) mm for mud crabs from fishing grid 1231, Adelaide River and 1536, Southern GOC for 2002 and 2003.

Area	Grid	Mean weight $\bar{w}$ (g)		Mean CW (mm)	
		2002	2003	2002	2003
Adelaide River	1231	812.4	772.9	157.1	156.5
Southern GOC	1536	682.2	647.4	151.0	150.0
<b>Annual mean entire fishery</b>		<b>714.4</b>	<b>660.2</b>	<b>152.9</b>	<b>151.6</b>

The estimated biomass of large crabs ( $\geq 120$ mm CW) for each study site was determined converting the size structure to total weight using the following formulae

$$\hat{b} = \hat{n} * \bar{w}$$

where

$\hat{b}$  = estimated biomass

$\hat{n}$  = estimated abundance

$\bar{w}$  = annual mean weight

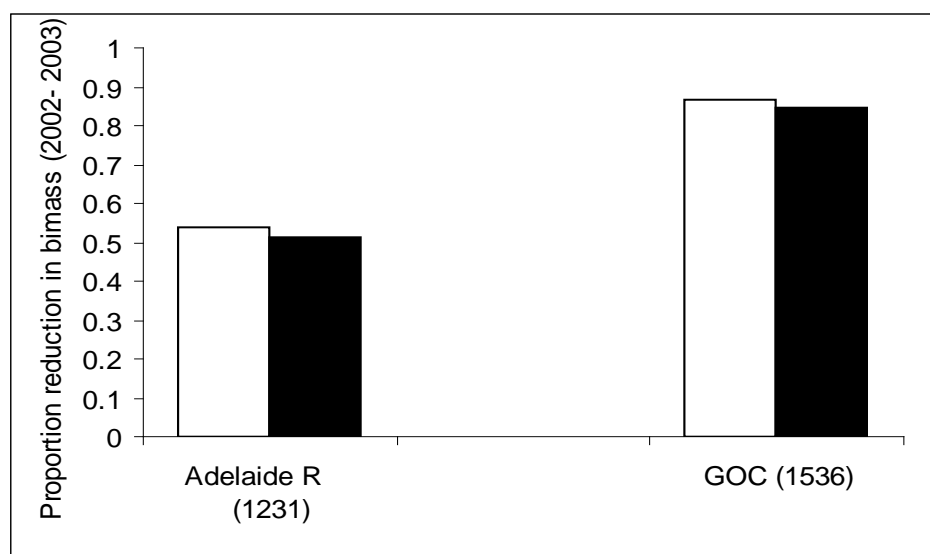
The exploitation rate was calculated as the percentage of the population removed (i.e. commercial catch) from the estimated biomass.

### 6.3.3 Results

Because of the differences in the proportion of area of foreshore mudflats in the Adelaide River and the GOC systems there are substantial differences in the contribution of mud crabs from each of these components to the catches in each fishing grid (Table 14).

**Table 14.** Comparison of the estimated biomass of larger crabs ( $\geq 120$  CW) and exploitation rates for fishing grids 1231 (Adelaide River) and 1536 (combined McArthur and Wearyan Rivers) between 2002 and 2003.

Grid	Year	Tidal River estimate n	Foreshore estimate n	Predicted (number)	Predicted Biomass (t)	Commercial catch (t)	Exploitation rate (% removed)
1231	2002	163686.69	64884.00	228570.69	185.69	38.07	20.5
	2003	78984.78	33164.90	112149.68	86.68	19.01	21.9
1536	2002	94189.16	812424.50	906613.66	618.49	288.55	46.7
	2003	13284.63	131424.56	144709.19	93.77	82.44	88.0



**Figure 34.** Comparison of the proportion reduction in estimated biomass of large crabs ( $\geq 120$  mm CW) between 2002 and 2003 for the Adelaide River (AR) and GOC. Declines in river (open histogram) and mudflat (solid histogram) estimates are equivalent within fishing grids.

The estimated biomass for mud crabs greater than 120 mm CW substantially declined from 2002 to 2003 and this decline was similar for both the mudflat and river components of the catch (Figure 34). The decline in abundance was substantially greater in the GOC (1536) than the Adelaide River (1231).

#### **6.3.4 Discussion**

The similarity in declines for both areas of the fishery in the Adelaide River and GOC is encouraging. Whilst this study cannot provide precise estimates of true abundance the results indicate that the methods tested here are feasible and show promise in tracking catch rates in the fishery.

Fishers are known to be methodical in harvesting all areas of the fishery and although the extent of mudflat and river areas vary between these two regions and the methods for estimating biomass in each habitat was substantially different, the similarity in the reduction in biomass for both habitats in each region further supports the use of the trapping web and mark-recapture estimators of abundance for this fishery.

The difference in exploitation rates between fishing regions and between years also reflects the socio-economic status of the fishery in both these regions. This increase in exploitation rate in the remote GOC is expected and can be attributed to the increased costs associated with working from a remote location driving fishers to continue to fish and attempt to recover outlays despite the reduced catch rates. As catch rates decline the increased costs may also provide some additional incentive to fish illegally. In contrast, the Adelaide River is only fished by a few fishers who tend to return to Darwin when catch rates are low and look for other work. With the low catch rates achieved in 2003 these fishers would be more likely to complete fishing trips earlier or skip fishing trips. There is also less incentive to fish illegally in this region through the opportunity for increased surveillance from the Darwin based Marine and Fisheries Enforcement Unit.

### **6.4 Comparison of metrics for the Northern Territory mud crab fishery**

#### **6.4.1 Introduction**

A range of metrics are commonly used as proxies for fish abundance during fishery assessments. These may include commercial catch, effort and catch rate data and fishery independent (FI) estimates including fishery independent catch rate data and abundance estimates. Changes in either catch or effort are usually used if only one is available. When both are available catch rate (catch/effort) is the preferred metric as it standardizes the changes in the catch against effort changes. Catch rate, or catch per unit of effort (CPUE) is more frequently used as a relative indicator of change in abundance. As CPUE is a function of the catchability and abundance, CPUE can change independently of abundance if catchability changes. In most fisheries catchability is considered constant between years although it can vary seasonally due to factors such as moulting, mating, water temperature (Zeigler *et al.* 2004). If catchability is considered to be equal between seasons then relative changes in abundance should be similar to relative changes in CPUE between seasons.

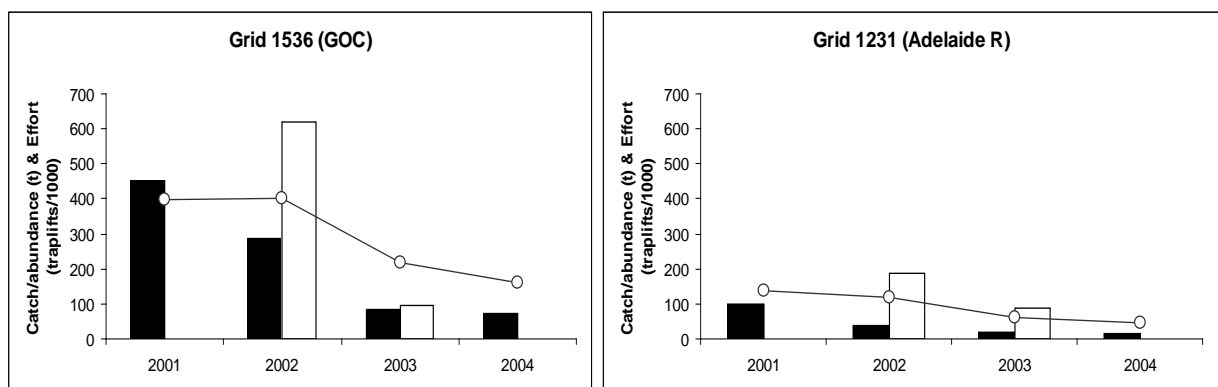
In some cases commercial CPUE has been found to be a poor indicator of fish abundance (Forrest and Pitcher, 2006). The authors cite a range of reasons including “changes in spatial distribution of the fish or the fishing fleet; changes in targeting practices; changes in catching power; the way in which the CPUE index is calculated (e.g., Walters 2003)”.

For *S. serrata* fisheries, the value of commercial and fishery independent CPUE analyses has proved cryptic due to numerous conflicting reports. Williams and Hill (1982) recommended the use of fishery independent CPUE in estimating *S. serrata* abundance over estimates obtained from mark-recapture data based on the observed unequal capture probability and the inability of the available models (Jolly- Seber stochastic model and the Manly and Parr model, 1968) to fit the data. In a later study Robertson (1989) assessed the factors affecting catches of *S. serrata* in baited traps and found that fishery independent CPUE was a poor index of abundance for trap-caught *S. serrata* due to the catch being asymptotic, reporting that CPUE decreased with soak time. Most recently in a 2006 assessment of the Northern Territory *S. serrata* fishery, Walters *et al.* (1997) determined that the use of commercial CPUE data was not informative for assessment due to fishers exhibiting non-random search behaviour, effectively moving to an unfished location as catch rates declined.

Considerable changes in commercial catch between years have been observed for this fishery and while it is impossible to be definitive regarding which metric or combinations of metrics are correct, this section compares a range of fishery dependent and independent metrics and discusses the merits of each.

## 6.4.2 Methods

When predicted biomass and commercial catch and effort for the two study sites are compared a similar pattern in reduced catch and effort is observed at both study sites (Figure 35).

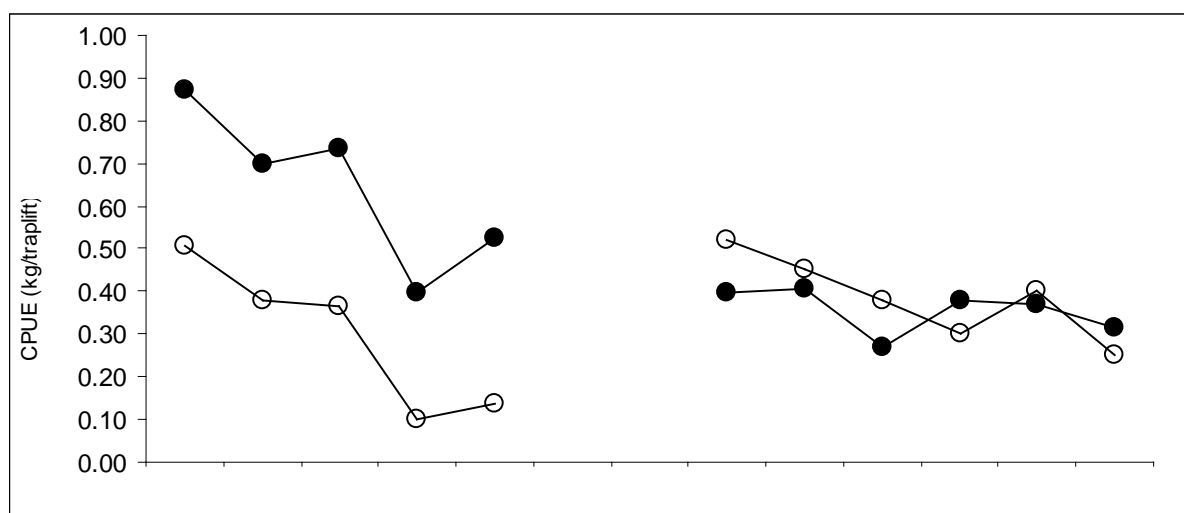


**Figure 35.** Comparison of the total commercial catch (solid histogram), predicted abundance (biomass) (open histogram) and commercial effort (open circle) for the fishing grids encompassing the two study sites 2002-2003.

For both fishing regions I compared the catch, effort and catch rate data from the commercial fishery as reported in the Northern Territory’s mandatory fishing logbooks. From this study I generated fishery independent catch rate data and abundance estimates. To standardise these different metrics, each metric is presented as the proportional reduction between the 2002 and 2003 fishing seasons.

At both the Adelaide River and GOC study sites fishery independent datasets were collected from areas commercially fished.

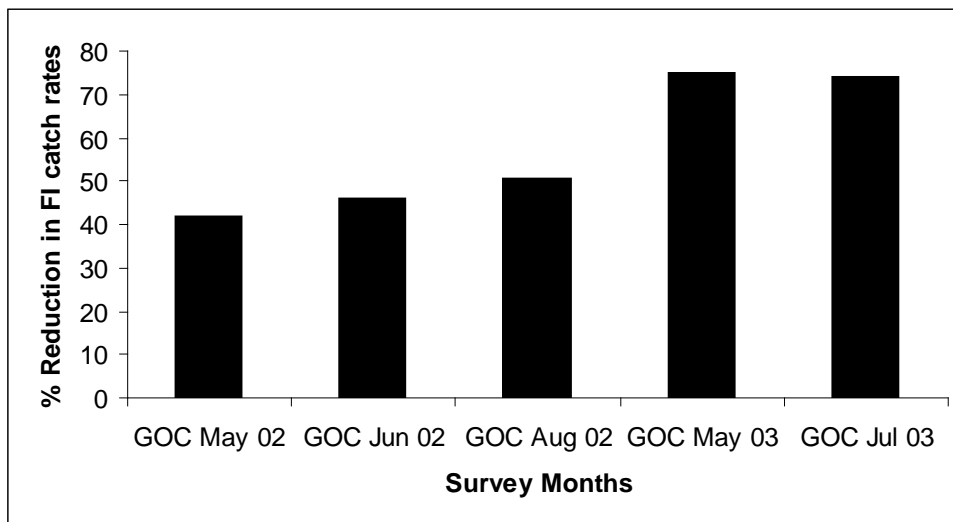
Commercial catch is validated against transport logs, whereas commercial fishing effort cannot be validated due to the remoteness of the fishing grounds. Reports from Northern Territory Fisheries Police, Marine and Fisheries Enforcement Unit indicate that use of excessive traps is the most common offence in the GOC region. If considerable unreported traps were in use, increases in catch would positively bias CPUE data.



**Figure 36.** Comparison of reported commercial (closed circles) and fishery independent (open circles) CPUE (kg/traplift) for the Adelaide River (AR) and GOC regions in 2002 & 2003.

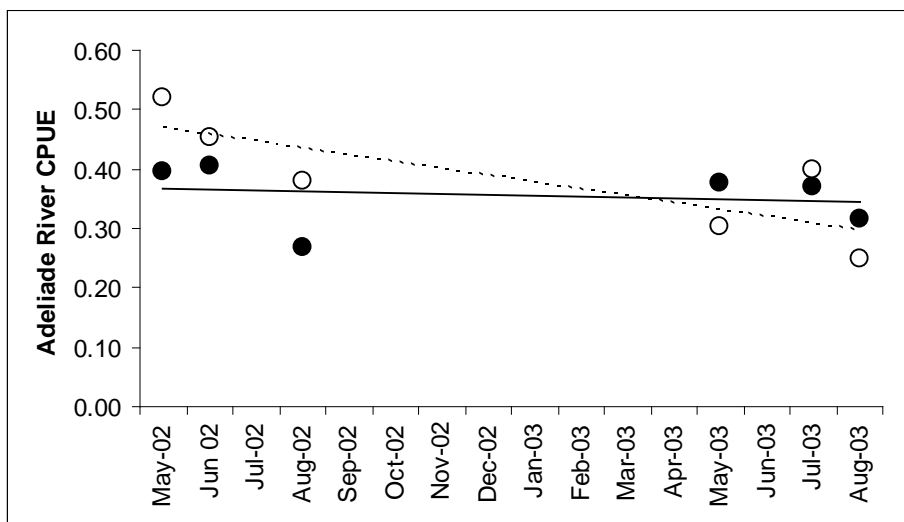
The commercial and fishery independent catch rates were strongly correlated for the datasets ( $r=0.98$ ).

The fishery independent catch rates were consistently lower than the commercial catch rate estimates in the GOC by approximately 40-50% during 2002 and this increased to approximately 75% for 2003 (Figure 37). The greater reduction in commercial CPUE when compared to the fishery independent CPUE in 2003 may be an indication of an increased use of excess traps in 2003, as reported by the Northern Territory Police, Marine and Fisheries Unit.



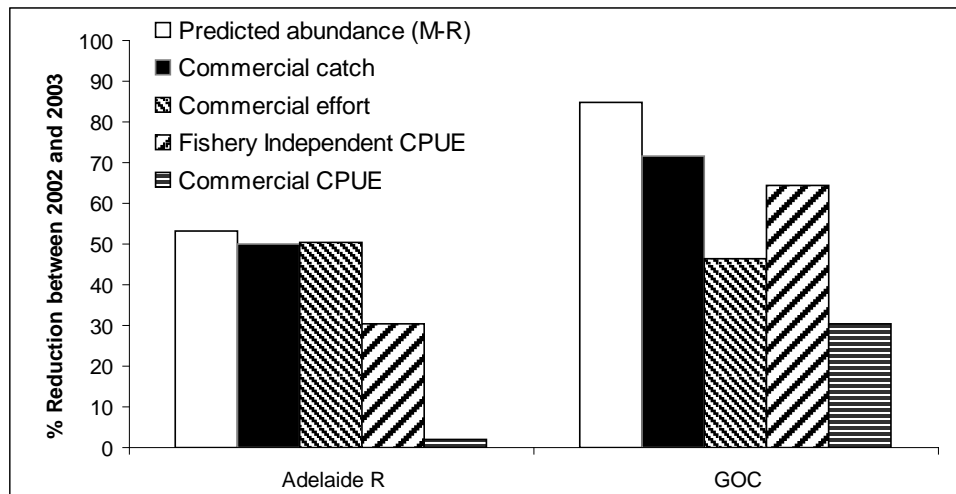
**Figure 37.** Percentage reduction in GOC fishery independent catches rates in comparison to commercial catch rates.

In contrast the fishery independent catch rate data from the Adelaide River was only moderately correlated with the commercial catch rate data ( $r=0.51$ ). Unlike the GOC, the commercial and fishery independent catch rate data for the Adelaide River were similar. However, trends between the two years were different with the fishery independent catch rate declining and the commercial catch rate remaining constant between fishing years (Figure 38).



**Figure 38.** Comparison of the commercial (closed circles and solid regression line  $r^2 = 0.04$ ) and fishery independent (open circles and dashed regression line  $r^2 = -0.64$ ) for Adelaide River surveys undertaken in 2002 and 2003.

Percentage changes in the fishery independent abundance and CPUE, commercial catch, commercial effort and commercial CPUE are presented in Figure 39. Although the rate of decline in the fishery independent CPUE (63.7%) was less than the abundance estimate (84.8%), both estimates clearly indicate substantial reductions in the abundance of crabs between 2003. In contrast commercial CPUE declined by only 46.8%. In the Adelaide River, fishery independent CPUE also declined at a lower rate than the abundance estimate although both declines were lower than respective estimates in the GOC. The commercial CPUE remained relatively stable between years.



**Figure 39.** Comparison of the percent reduction of commercial catch, predicted catch, commercial fishing effort, commercial CPUE and Fishery Independent CPUE between 2002 and 2003 at each site.

## 6.5 Discussion

As the total catch in the Adelaide River and GOC had declined by 52% and 72% respectively, the reduced abundances and CPUE estimates from the fishery independent metrics appear more realistic as indicators of the status of the stocks than commercial catch and effort data. Ancillary data from the compliance section of the NT marine police provide further anecdotal support for interpretation of these findings. The Marine police report that the use of illegal traps is the main compliance issue in the GOC and this would result in declines in commercial CPUE data being underestimated. In contrast, the Adelaide River region is less isolated and readily accessible to the marine police. Because of the lower catches from this region, the majority of fishers are part-time with income derived from other businesses. Fishers operating in the Adelaide River would be expected to reduce effort when catch rates start to decline. The hyper-stability of catch rates for this region would therefore be attributed to opportunity costs and income derived from other ventures. Fishers in the GOC do not have similar opportunity costs and fishing is their primary livelihood.

The lower fishery independent CPUE compared to the abundance estimates in both river systems would suggest that catchability would also have been impacted. Lower abundance and density of crabs would be expected to lower catchability as fewer crabs would be available for capture per unit of fishing gear.



## 7. DISCUSSION

### 7.1 Estimating mud crab abundance

The need to achieve reliable stock density estimates of commercially fished populations are a fundamental requirement for effective and sustainable fishery management. Worldwide a variety of assessment methods have been examined for crustacean fisheries. Methods such as trawling (Weinberg *et al.* 2004), visual techniques such as transect surveys (Taggart *et al.* 2004), videography (Melville-Smith 1986) and burrow density/occupancy rates (Barnes *et al.* 2002) have been reported as useful tools in crustacean fisheries assessment. However the turbid waters of tropical Australia rule out any investigation using visual methods. Williams and Hill (1982) after considerable efforts in assessing methods for mud crab abundance estimation conclude “The high turbidity of estuarine and inshore waters which this species inhabits precludes diver counts and few crabs are caught in trawls, probably because they are able to avoid them by burrowing and swimming”. Also diving in the highly turbid inshore waters prevalent in northern Australia may prove a short term occupation due to the presence of large predators, especially crocodiles.

A number of studies investigating wild mud crab population abundance have been documented. Hill (1975) using baited box traps and the assumption of a linear relationship between catch-per-unit-effort (CPUE) and population density, estimated mud crab abundance in the Kleinmond Estuary, South Africa, to be 0.806 crabs per 100m<sup>2</sup>. Hill (1979a) later also provided abundance estimates for three areas of the St Lucia estuarine system, South Africa, as 0.156 crabs per 100m<sup>2</sup> (South Lake), 0.037 crabs per 100m<sup>2</sup> (North Lake) and 0.339 crabs per 100m<sup>2</sup> (Narrows).

Williams and Hill (1982) identified baited traps as the most practical method of sampling *S. serrata* and used baited crab traps to gain population estimates for mud crabs in an estuarine area of Moreton Bay, Queensland, comparing the use of CPUE and mark-recapture methods. They recommended the use of fishery independent CPUE data as a measure of relative abundance of adult crabs (after making allowance for water temperature and moult incidence) over the use of mark recapture techniques. They identified cost and failure to meet the equal catchability assumption (see Chapter 2.8.1 assumption 4) as key factors precluding the use of mark-recapture methods in estimating population size of *S. serrata*. Alternately Robertson (1989) assessed the factors affecting catches of *S. serrata* in baited traps and found that CPUE was a poor index of abundance for trap-caught *S. serrata* due to catch being asymptotic, reporting that CPUE decreased with soak time. Walters *et al.* (1997) determined that assessment methods based on commercial catch-rate data and CPUE based models were not informative and should not be used in assessment of the Northern Territory mud crab fishery. This conclusion was based on finding non-randomness in the spatial pattern of fishing effort, where fishermen appeared to systematically deplete local areas then shift effort to new areas over the fishing season, so as to maintain hyperstability in catch per effort.

Robertson and Piper (1991) tested newly developed mark-recapture models designed to relax some of the more rigid assumptions of mark-recapture theory and permit heterogeneous capture probabilities. A suite of models were chosen (Petersen 1896; Burnham and Overton 1978; Burnham and Overton 1979; Chao 1987) to estimate the population size of *S. serrata* in two closed estuaries in South Africa. Of these, the Chao model provided the most reliable estimates of population size. This model permits the estimation of population size using mark-recapture data with unequal catchability. These and other more recent and advanced models now form the basis of the two general approaches for estimating animal densities. These are

line (or point) transect surveys commonly recommended for terrestrial animals and birds (Distance sampling), and mark-recapture techniques. The work undertaken in this study provides two new study designs applied for the first time in a fishery context.

Prior to estimating mud crab abundance a series of preliminary investigations were completed. High levels of variation in historical catch and effort data lead to examination of some of the biological and environmental influences on mud crab catchability.

*Scylla serrata* is known to be an aggressive predator and the influence of crab size on catchability was investigated. Results indicated that there is an interaction between larger and smaller mud crabs, with the presence of larger crabs influencing the catchability of smaller crabs. These results highlight the need for a cautionary approach in estimating abundance of small crabs (<120mm CW) from traps. Further research is required to determine methods for standardising catch rates of smaller crabs against the catch rates or abundance of larger crabs. For this study we split crabs into two size bins (<120mm and  $\geq 120$ mm) and selected mark-recapture models that allowed for time dependent changes in each of the size classes.

Finally, an important assumption of closed population mark-recapture and depletion analyses is that the population remains closed to immigration or emigration during the study period. I examined the movement patterns of 351 tagged crabs within a 2km stream study site. 100 baited traps were set at 20m intervals along the stream and 88.1% of tagged crabs were recaptured within 100m of their previous capture location. I then examined for two types of movement patterns, movement of tagged crabs within the buffer zones and from buffer zones into the depletion zone over the four day period immediately after tagging. The change in proportion of crabs, recaptured in the buffer zones and that moved from the buffer zones into the depletion zone, was not significant. These results infer that the removal of animals from the depletion zone does not encourage large scale immigration of crabs from the buffer zones supporting the assumption of a closed population during the study period.

Comparison of results obtained from the mark-recapture and depletion studies resulted in similar numbers of captures, abundance estimates and confidence intervals of similar magnitude, demonstrating no real preference for either design. Declining numbers in catch were observed at each study site in 2003 and the design and analyses models continued to provide acceptable results despite very low numbers of captures and recaptures.

Parmenter *et al.* (2003) evaluated the accuracy of various models in estimating small mammal density by conducting mark-recapture and trapping web studies on known densities of enclosed rodent populations. When basic assumptions were met the use of web based analytical methods to estimate abundance were recommended over grid based approaches due to the sound theoretical basis of distance sampling techniques.

Preliminary trapping web analyses provided some evidence that a reduction in the size of the area sampled and small numbers of captures/recaptures, and excessive movement may fail some of the underlying analysis assumptions. Buckland *et al.* (2001) in assessing the use of trapping web data warns that over estimation of a population's density may occur if the species of interest displays excessive movement near the centre of the web. This can be problematic particularly when using baited traps and when trap spacing is too small. "If animals tend to move in home ranges that are small relative to the size of the web and trap spacing then the trapping web is likely to perform well. Alternatively if animals move somewhat randomly over wide areas in relation to the size of the web and trap interval chosen then overestimation may be substantial."

Two analysis methods for trapping web data were tested during this study. One (DISTANCE) uses only first capture information and the other (DENSITY) uses both capture and recapture information. Intuitively an analysis using all the available information would prove more robust; however DISTANCE is the standard, well-known analysis method while DENSITY is a new and relatively untested method. Efford *et al.* (2005) tested both methods of analysis on a population of brushtail possums (*Trichosurus vulpecula*) in New Zealand. The true density of the total possum population was determined through exhaustive removal. Results indicated that the estimate obtained using inverse prediction (DENSITY software) of 1.88 possums per hectare was consistent with the removal estimate (2.27/ha), whereas estimates from trapping webs using DISTANCE software were positively biased (6.5 to 8.0/ha, depending on the model used).

On review estimates of abundance from the trapping web studies gained from the two analysis methods were of similar magnitude, generally followed similar trends and provided feasible estimates of mud crab density, with DISTANCE results demonstrating better precision. This provides support for the use of this trapping web methodology in the marine environment.

When relative abundance estimates from the two Northern Territory study sites were scaled by the estimated area of habitat in each corresponding fishing grid, results followed commercial catch trends. The GOC remains the most significant region for the Northern Territory mud crab fishery responsible for 77% of the total Northern Territory catch in 2003. Comparison of the reported commercial catch to the estimated abundance of mud crabs in 2003 confirmed the high levels of exploitation 70-90% reported by Walters *et al.* (1997). This study estimated that the GOC fishery exploitation rate was around 88% of adult biomass in 2003. An exploitation rate of around 21% of adult biomass was estimated for the Adelaide River region in 2003.

The two Northern Territory study areas demonstrated differences in abundance patterns between habitat types. Over the two-year study period, the majority of crabs caught at the Adelaide River study site (89%) were captured within the creek habitat. This could be the result of high tidal movement in this area, encouraging crabs to seek refuge in sheltered creeks. This pattern was not evident for the GOC, where tidal range is considerably smaller and similar patterns in abundance were observed in both the Gulf creeks (48%) and Gulf foreshore flat (52%) habitats. A similar difference in habitat preference by gender was evident for both study sites. Male mud crabs dominated the creek catches at both the Adelaide (63% male) and Gulf (73% male) river sites. At the two foreshore sites, females dominated the catch contributing 62% of the Adelaide River foreshore catch and 68% of the total Gulf catch.

During the two years of this study, commercial catch reported from within the fishing grids containing each study site, declined. A range of indices were examined for similar patterns in decline. As noted by Walters *et al.* (1997) commercial fishing catch rates did not reflect trends in abundance. The fishery independent catch rates reflected some of the change between years, however the changes in predicted abundance estimates obtained from mark-recapture methods best reflected the trends in commercial catch.

## 8.0 CONCLUSION

The tagging study designs developed during this research and the analyses methods outlined in this document have not previously been applied to a fishery. While the results demonstrate some variability within year; the study designs provide biologically reasonable estimates of mud crab biomass that compare favourably with fishery data. These results provide good support for the further development and testing of the model, *Abundance = area of critical habitat x density of animals per unit of habitat*. Study site selection and an understanding of animal movement, animal behaviour and seasonal patterns are important factors for consideration when planning mud crab abundance surveys. The methods presented in this thesis including depletion and a non depletion component that enable factors such as changes in selectivity to be accounted for. The study design also appears to meet the critical analysis assumption of closure.

The methods of assessing mud crab abundance provides an additional tool for fishery assessment, particularly as in this case, when a component of the catch and effort data cannot be validated. High levels of unreported effort in the Northern Territory hamper effective evaluation of fishing activity and the additional information gained from this work has proved important in the most recent assessment of the fishery. Additional and important benefits to be gained from the intensive tagging studies undertaken during this project are a better understanding of the mud crab's population dynamics and improved estimates of various parameters used for fishery assessment such as growth, recruitment, and selectivity. This study also provides insights into a dynamic and complex system that has historically proven difficult to model. For the first time differences in habitat preference were observed between region and sex.

The application of the density and distance methods are novel in a fisheries context and deserve further investigation, particularly in the evaluation of analysis assumptions to demonstrate the potential of these analysis techniques.

Fisheries researchers have traditionally put little effort into estimation of density, instead focussing on abundance and biomass estimation. Density becomes an important measure where biomass is linked to a habitat type and traditional approaches such as transects counts are limited. The value of this approach is in the integration and understanding of spatial management and ecological research in the adoption of ecosystem based management of natural resources.

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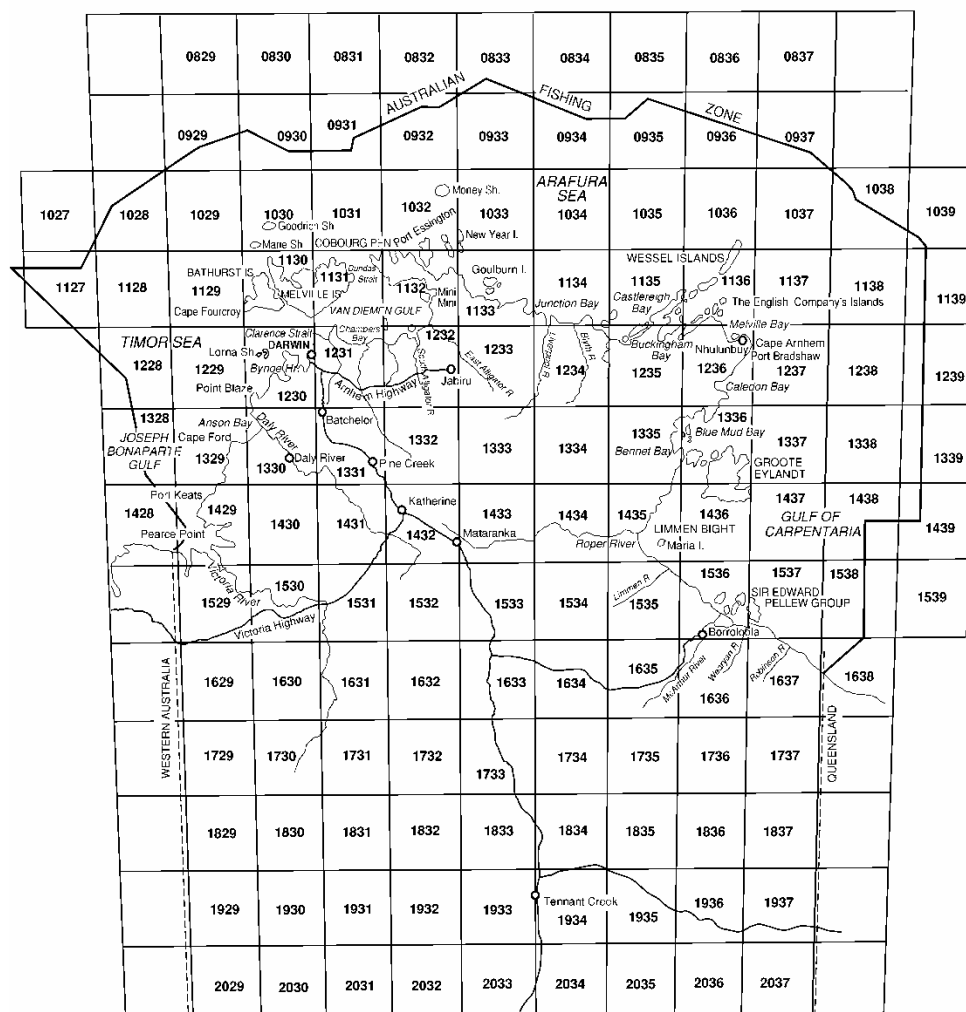
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# Appendix 1. Northern Territory Reference Fishing Grid

## NORTHERN TERRITORY OF AUSTRALIA – FISHING GRIDS REFERENCE



**Appendix 2 Table 15.** Comparison of depletion and mark recapture estimates of crabs ( $\hat{N}$ ) <120 mm and  $\geq 120$  mm CW, standard error for buffer 1 (B1), depletion (D) and buffer 2 (B2).

Site	Date	zone	# marked		model	$\hat{N}$ <120	s.e	D/km <sup>2</sup>	$\hat{N}$ $\geq 120$	s.e	D/km <sup>2</sup>
			<120	$\geq 120$							
AR	May-02	B1	27	81	p(.)c(g*t)	40.91	8.61	2727	122.72	23.26	8181
		D	91	79	p(small t*+size), lge*size	126.78	36.36	4226	80.45	0.76	2682
		B2	49	84	p(.)c(g)	60.51	5.95	4034	103.74	9.31	6916
AR	Jul-02	B1	30	72	p(.) c(g)	39.35	5.67	2623	94.44	12	6296
		D	143	63	p(size)c=0	250.97	48.86	8366	67.75	3.01	2258
		B2	83	42	p(.) c(g)	117.22	17.14	7815	59.32	9.3	3955
AR	Aug-02	B1	36	58	p(.) c(g)	54.68	11.79	3645	88.09	18.24	5873
		D	115	41	p(size) c=0	164.03	29.94	5468	41.65	1.06	1388
		B2	81	19	p(.) c(g)	100.93	10.5	6729	23.67	3.25	1578
AR	May-03	B1	8	36	p(.) c(g)	8.60	0.9	573	38.71	2.5	2581
		D	41	81	p(size)c=0	50.77	6.18	1692	88.62	4.37	2954
		B2	21	46	p(.) c(g)	24.52	2.8	1635	53.70	5.19	3580
AR	Jul-03	B1	29	30	p(.) c(.)	32.95	2.48	2197	33.05	2.54	2203
		D	74	46	p(small*t+size, lge) c=0	77.05	3.66	2568	47.10	1.07	1570
		B2	24	15	p(g*t)c(t)	50.88	0	3392	47.07	0	3138
AR	Sep-03	B1	30	15	p(.) c(.)	32.36	2.23	2157	16.18	1.4	1079
		D	48	24	p(size)c=0	54.21	4.4	1807	25.15	1.34	838
		B2	20	14	p(g) c(.)	29.17	11.47	1945	14.06	0.26	937
GOC	May-02	B1	10	101	p(.) c(g*t)	17.98	6.14	514	181.64	50.32	5190
		D	19	123	p(small*t, lge) c=0	20.99	0	300	215.75	53.73	3082
		B2	8	41	p(.) c(g)	12.68	4.67	362	65.01	20.42	1857
GOC	Jun-02	B1	21	98	p(.) c(t)	29.54	5.2	844	137.83	20.1	3938
		D	148	153	p(size)c=0	107.78	38.55	1540	179.20	11.29	2560
		B2	13	69	p(.) c(t)	19.69	5.1	563	104.48	22.55	2985
GOC	Aug-02	B1	7	35	p(.) c(.)	8.69	1.89	248	43.44	6.88	1241
		D	17	54	p(size)c=0	27.90	9.94	399	60.31	4.57	862
		B2	3	57	p(.) c(.)	3.24	0.53	93	61.64	3.48	1761
GOC	May-03	B1	3	15	p(.) c(t)	3.50	0.93	100	17.51	3.16	500
		D	14	10	p(size)c=0	15.95	2.9	228	10.45	0.89	149
		B2	2	7	p(.) c(g)	2.00	0.07	57	7.02	0.13	201
GOC	Jun-03	B1	6	26	p(.) c(.)	6.16	0.43	176	26.67	1.02	762
		D	14	29	p(small*t+size, lge (.) c=0	14.00	0.004	200	29.19	0.49	417
		B2	1	6	p(.) c(.)	1.04	0.21	30	6.24	0.65	178

**Table 16.** Trapping web density estimates (km<sup>2</sup>) for all crabs for the two study areas. DISTANCE analyses results, ΔAIC and AIC model selection, Density/km<sup>2</sup> and upper and lower confidence levels. DENSITY analyses results D/km<sup>2</sup> and capture probability (g0) magnitude and spatial scale (home range) sigma (m). See Chapter 6.2.2.

DISTANCE all crab								DENSITY all crab									
Site	Date	Detection Model	ΔAIC	AIC	D/km <sup>2</sup>	seD	D CV	Detection	N-hat	seN	p hat	d bar	sed	D/km <sup>2</sup>	seD	g0	Sigma
Adelaide	May-02	uniform cosine	0	209.06	2552.50	482.38	0.189	Uniform	281.88	23.86	0.099	103.4	12.8	1164.26	722	0.002	262.68
		half normal cosine	0	209.15	2455.80	524.31	0.214	Halfnormal	281.88	23.86	0.099	103.4	12.8	1526.47	538	0.003	129.79
Adelaide	Jul-02	uniform cosine	0.353	65.711	719.86	262.55	0.365	Uniform	95.47	13.95	0.126	86.1	18.3	634.51	268	0.005	138.37
		half normal cosine	0	65.358	675.27	265.40	0.393	Halfnormal	95.47	13.95	0.126	86.1	18.3	NA	NA	NA	NA
Adelaide	Aug-02	uniform cosine	0	61.341	363.08	81.19	0.459	Uniform	66.66	11.52	0.129	66.6	15	655.27	229	0.008	91.81
		half normal cosine	2	63.341	363.15	166.50	0.224	Halfnormal	66.66	11.52	0.129	66.6	15	619.56	251	0.015	49.14
Adelaide	May-03	uniform cosine	0	42.175	428.89	114.63	0.548	Uniform	73.1	12.3	0.090	94.1	31.8	617.03	513	0.002	186.59
		half normal cosine	2	44.175	428.98	235.09	0.267	Halfnormal	73.1	12.3	0.090	94.1	31.8	NA	NA	NA	NA
Wearyan	May-02	uniform cosine	0.67	789.96	7194.20	937.48	0.13	Uniform	847.21	41.38	0.093	112.9	8.6	3279.19	392	0.001	408.19
		half normal cosine	0	789.29	7225.50	960.49	0.133	Halfnormal	847.21	41.38	0.093	112.9	8.6	3311.73	498	0.002	328.29
Wearyan	Aug-02	uniform cosine	0	201.42	1060.80	236.11	0.223	Uniform	343.83	26.29	0.072	103.8	34.3	NA	NA	NA	NA
		half-normal cosine	0.32	201.74	986.80	225.86	0.229	Halfnormal	343.83	26.29	0.072	103.8	34.3	NA	NA	NA	NA
Wearyan	Aug-02	uniform cosine	0.95	432.66	3251.30	660.37	0.203	Uniform	467.38	30.82	0.109	66.7	9.6	4880.47	760	0.001	92.94
		half-normal cosine	0	431.71	4103.80	970.17	0.236	Halfnormal	467.38	30.82	0.109	66.7	9.6	4769.81	807	0.012	48.84
Wearyan	May-03	uniform cosine	0.06	144.61	1442.10	329.26	0.228	Uniform	160.31	18.11	0.104	96.9	19.2	926.61	564	0.003	188.43
		half normal cosine	0	144.55	1398.10	359.57	0.257	Halfnormal	160.31	18.11	0.104	96.9	19.2	891.94	346	0.005	103.64
Wearyan	Jun-03	uniform cosine	0.64	210.45	2901.00	758.92	0.262	Uniform	234.82	21.84	0.089	67.8	13.8	3572.64	1058	0.003	94.28
		half normal cosine	0	209.81	3854.90	1183.10	0.307	Halfnormal	234.82	21.84	0.089	67.8	13.8	3523.69	1245	0.006	50.77
Wearyan	Aug-03	uniform cosine	0	275.74	7508.50	2189.30	0.292	Uniform	356.28	26.8	0.083	77.9	21.9	5425.37	934	0.002	118.23
		half normal cosine	0.08	275.82	5869.20	1456.00	0.249	Halfnormal	356.28	26.8	0.083	77.9	21.9	5727.00	3433	0.004	61.9

**Table 17.** Trapping web density estimates (km<sup>2</sup>) for crabs  $\geq 120$  mm for the two study areas. DISTANCE analyses results,  $\Delta$ AIC and AIC model selection, Density/km<sup>2</sup> and upper and lower confidence levels. DENSITY analyses results D/km<sup>2</sup> and capture probability (g0) magnitude and spatial scale (home range) sigma (m) see Chapter 6.2.2.

DISTANCE crab $\geq 120$								Density crab $\geq 120$									
Site	Date	Detection Model	$\Delta$ AIC	AIC	D/km <sup>2</sup>	seD	D CV	Detection	N-hat	seN	p hat	d bar	sed	D/km <sup>2</sup>	seD	g0	Sigma
Adelaide	May-02	uniform cosine	0.1	116.24	1147.4	340.81	0.297	Uniform	175.7	18.93	0.112	116.6	12.8	482.22	89.6	0.002	547.96
		half normal cosine	0	116.14	1062.5	321.3	0.3024	Halfnormal	175.7	18.93	0.112	116.6	12.8	NA			
Adelaide	Jul-02	uniform cosine	0.04	42.74	673.85	230.46	0.342	Uniform	79.87	12.75	0.120	104.5	21.3	312.20	212.0	0.005	141.98
		half normal cosine	0	42.70	697.52	308.84	0.4428	Halfnormal	79.87	12.75	0.120	104.5	21.3	356.81	310.8	0.004	159.21
Adelaide	Aug-02	uniform cosine	0	34.503	217.85	62.888	0.2887	Uniform	43.14	8.72	0.144	66.2	16.6	435.54	174.4	0.009	91.36
		half normal cosine	2.00	36.504	217.89	128.97	0.5919	Halfnormal	43.14	8.72	0.144	66.2	16.6	397.40	160.6	0.016	49.44
Adelaide	May-03	uniform cosine	0	32.171	336.99	101.61	0.3015	Uniform	60.88	11.25	0.095	94.1	31.8	468.85	588.3	0.002	184.02
		half normal cosine	2.00	34.172	337.05	208.38	0.6182	Halfnormal	60.88	11.25	0.095	94.1	31.8	NA	NA	NA	NA
Wearyan	May-02	uniform cosine	0.5	720.68	6281.2	881.56	0.1403	Uniform	772.4	39.52	0.095	114.7	8.5	2518.45	329.9	0.001	537.88
		half normal cosine	0	720.18	6273.7	893.02	0.1423	Halfnormal	772.4	39.52	0.095	114.7	8.5	NA	NA	NA	NA
Wearyan	Aug-02	uniform cosine	0	143.07	428.52	63.88	0.1491	Uniform	280.79	23.77	0.073	103.8	34.3	NA	NA	NA	NA
		half-normal cosine	0.4	143.47	590.09	166.19	0.2816	Halfnormal	280.79	23.77	0.073	103.8	34.3	NA	NA	NA	NA
Wearyan	Aug-02	uniform cosine	1.63	332.67	3232.7	918.32	0.2841	Uniform	372.48	27.5	0.114	70.3	10.1	3500.07	535.9	0.006	100.7
		half-normal cosine	0	331.04	3031.7	831.19	0.2742	Halfnormal	372.48	27.5	0.114	70.3	10.1	3313.51	640.3	0.012	54.02
Wearyan	May-03	uniform cosine	0	64.367	299.06	66.873	0.2236	Uniform	75.45	11.8	0.135	110.7	19.2	283.44	91.6	0.003	364.68
		half normal cosine	0.654	65.021	460.27	189.19	0.411	Halfnormal	75.45	11.8	0.135	110.7	19.2	296.08	100.3	0.004	207.88
Wearyan	Jun-03	uniform cosine	0	99.776	767.18	254.41	0.3316	Uniform	105.17	14.81	0.112	64.2	14.3	1119.80	393.8	0.007	89.46
		half normal cosine	0.04	99.819	1320.1	686.91	0.5203	Halfnormal	105.17	14.81	0.112	64.2	14.3	1120.46	364.3	0.013	47.27
Wearyan	Aug-03	uniform cosine	0	108.61	508.41	87.191	0.1715	Uniform	168.24	18.56	0.087	115.8	33	679.75	241.0	0.001	996.18
		half normal cosine	1.48	110.09	629.8	209.56	0.3327	Halfnormal	356.28	26.8	0.083	77.9	21.9	997.08	624.1	0.001	3171.43